6 FIELD MEASUREMENT METHODS AND INSTRUMENTATION

6.1 Introduction

Measurement is used in MARSSIM to mean 1) the act of using a detector to determine the level or quantity of radioactivity on a surface or in a sample of material removed from a media being evaluated, or 2) the quantity obtained by the act of measuring. Three methods are available for collecting radiation data while performing a survey—direct measurements, scanning, and sampling. This chapter discusses scanning and direct measurement methods and instrumentation. The collection and analysis of media samples are presented in Chapter 7. Information on the operation and use of individual field and laboratory instruments is provided in Appendix H. Quality assurance and quality control (QA/QC) are discussed in Chapter 9.

Total surface activities, removable surface activities, and radionuclide concentrations in various environmental media (*e.g.*, soil, water, air) are the radiological parameters typically determined using field measurements and laboratory analyses. Certain radionuclides or radionuclide mixtures may necessitate the measurement of alpha, beta, and gamma radiations. In addition to assessing each survey unit as a whole, any small areas of elevated activity should be identified and their extent and activities determined. Due to numerous detector requirements, no single instrument (detector and readout combination) is generally capable of adequately measuring all of the parameters required to satisfy the release criterion or meet all the objectives of a survey.

Selecting instrumentation requires evaluation of both site and radionuclide specific parameters and conditions. Instruments should be stable and reliable under the environmental and physical conditions where they are used, and their physical characteristics (size and weight) should be compatible with the intended application. The instrument and measurement method should be able to detect the type of radiation of interest, and should, in relation to the survey or analytical technique, be capable of measuring levels that are less than the derived concentration guideline level (DCGL). Numerous commercial firms offer a wide variety of instruments appropriate for the radiation measurements described in this manual. These firms can provide thorough information regarding capabilities, operating characteristics, limitations, *etc.*, for specific equipment.

If the field instruments and measurement methods cannot detect radiation levels below the DCGLs, laboratory methods discussed in Chapter 7 are typically used. A discussion of detection limits and detection levels for some typical instruments is presented in Section 6.7. There are certain radionuclides that will be essentially impossible to measure at the DCGLs *in situ* using current state-of-the-art instrumentation and techniques because of the types, energies, and abundances of their radiations. Examples of such radionuclides include very low energy, pure beta emitters such as ^{3}H and ^{63}Ni and low-energy photon emitters such as ^{55}Fe and ^{125}I. Pure alpha emitters dispersed in soil or covered with some absorbing layer may not be detectable because alpha radiation will not penetrate through the media or covering to reach the detector. A

common example of such a condition would be ^{230}Th surface contamination, covered by paint, dust, oil, or moisture. NRC report NUREG-1507 (NRC 1997a) provides information on the extent to which these surface conditions may affect detection sensitivity. In circumstances such as these, the survey design will usually rely on sampling and laboratory analysis to measure residual activity levels.

6.2 Data Quality Objectives

The third step of the Data Quality Objectives (DQO) Process involves identifying the data needs for a survey. One decision that can be made at this step is the selection of direct measurements for performing a survey or deciding that sampling methods followed by laboratory analysis are necessary.

6.2.1 Identifying Data Needs

The decision maker and the survey planning team need to identify the data needs for the survey being performed, including the:

- type of measurements to be performed (Chapter 5)
- radionuclide(s) of interest (Section 4.3)
- number of direct measurements to be performed (Section 5.5.2)
- area of survey coverage for surface scans based on survey unit classification (Section 5.5.3)
- type and frequency of field QC measurements to be performed (Section 4.9)
- measurement locations and frequencies (Section 5.5.2)
- standard operating procedures (SOPs) to be followed or developed (Chapter 6)
- analytical bias and precision (*e.g.*, quantitative or qualitative) (Appendix N, Section N.6)
- target detection limits for each radionuclide of interest (Section 6.4)
- cost of the methods being evaluated (cost per measurement as well as total cost) (Appendix H)
- necessary turnaround time
- specific background for the radionuclide(s) of interest (Section 4.5)
- derived concentration guideline level (DCGL) for each radionuclide of interest (Section 4.3)
- measurement documentation requirements
- measurement tracking requirements

Some of this information will be supplied by subsequent steps in the DQO process, and several iterations of the process may be needed to identify all of the data needs. Consulting with a health physicist or radiochemist may be necessary to properly evaluate the information before deciding

between direct measurements or sampling methods to perform the survey. Many surveys will involve a combination of direct measurements and sampling methods, along with scanning techniques, to demonstrate compliance with the release criterion.

6.2.2 Data Quality Indicators

The data quality indicators identified as DQOs in Section 2.3.1 and described in Appendix N should be considered when selecting a measurement method (*i.e.*, scanning, direct measurement, sampling) or a measurement system (*e.g.*, survey instrument, human operator, and procedure for performing measurements). In some instances, the data quality indicator requirements will help in the selection of a measurement system. In other cases, the requirements of the measurement system will assist in the selection of appropriate levels for the data quality indicators.

6.2.2.1 Precision

Precision is a measure of agreement among replicate measurements of the same property, under prescribed similar conditions (ASQC 1995). Precision is determined quantitatively based on the results of replicate measurements (equations are provided in EPA 1990). The number of replicate analyses needed to determine a specified level of precision for a project is discussed in Section 4.9. Determining precision by replicating measurements with results at or near the detection limit of the measurement system is not recommended because the measurement uncertainty is usually greater than the desired level of precision. The types of replicate measurements applied to scanning and direct measurements are limited by the relatively uncomplicated measurement system (*i.e.*, the uncertainties associated with sample collection and preparation are eliminated). However, the uncertainties associated with applying a single calibration factor to a wide variety of site conditions mean these measurements are very useful for assessing data quality.

- Replicates to Measure Operator Precision. For scanning and direct measurements, replicates to measure operator precision provide an estimate of precision for the operator and the Standard Operating Procedure (SOP) or protocol used to perform the measurement. Replicates to measure operator precision are measurements performed using the same instrument at the same location, but with a different operator. Replicates to measure operator precision are usually non-blind or single-blind measurements.

- Replicates to Measure Instrument Precision. For scanning and direct measurements, replicates to measure instrument precision provide an estimate of precision for the type of instrument, the calibration, and the SOP or protocol used to perform the measurement. Replicates to measure instrument precision are measurements performed by the same operator at the same location, but with a different instrument. Replicates to measure instrument precision are usually non-blind or single-blind measurements.

For many surveys a combination of instrument and operator replicates are used to provide an estimate of overall precision for both scanning and direct measurements. Replicates of direct measurements can be compared with one another similar to the analytical results for samples. Results for scanning replicates may be obtained by stopping and recording instrument readings at specific intervals during the scanning survey (effectively performing direct measurements at specified locations). An alternative method for estimating the precision of scanning is to evaluate the effectiveness of the scanning survey for identifying areas of elevated activity. The results of scanning are usually locations that are identified for further investigation. A comparison of the areas identified by the replicate scanning surveys can be performed either quantitatively (using statistical methods) or qualitatively (using professional judgment). Because there is a necessity to evaluate whether the same number of locations were identified by both replicates as well as if the identified locations are the same, there is difficulty in developing precision as a DQO that can be evaluated.

6.2.2.2 Bias

Bias is the systematic or persistent distortion of a measurement process that causes error in one direction (EPA 1997a). Bias is determined quantitatively based on the measurement of materials with a known concentration. There are several types of materials with known concentrations that may be used to determine bias for scans and direct measurements.

- Reference Material. Reference material is a material or substance one or more of whose property values are sufficiently homogeneous and well established to be used for the calibration of an apparatus, the assessment of a measurement method, or for assigning values to materials (ISO 1993). A certified reference material is reference material for which each certified property value is accompanied by an uncertainty at a stated level of confidence. Radioactive reference materials may be available for certain radionuclides in soil (*e.g.*, uranium in soil), but reference building materials may not be available. Because reference materials are prepared and homogenized as part of the certification process, they are rarely available as double-blind samples. When appropriate reference materials are available (*i.e.*, proper matrix, proper radionuclide, proper concentration range) they are recommended for use in determining the overall bias for a measurement system. For scanning and direct measurements a known amount of reference material is sealed in a known geometry. This known material is measured in the field using a specified protocol (*e.g.*, specified measurement time at a specified distance from the reference material) to evaluate the performance of the instrument only.

- Performance Evaluation (PE) Samples. PE samples are used to evaluate the bias of the instrument and detect any error in the instrument calibration. These samples are usually prepared by a third party, using a quantity of analyte(s) which is known to the preparer but unknown to the operator, and always undergo certification analysis. The analyte(s)

used to prepare the PE sample is the same as the analyte(s) of interest (EPA 1991g). PE samples are recommended for use in determining bias for a measurement system when appropriate reference materials are not available. PE samples are equivalent to matrix spikes prepared by a third party that undergo certification analysis and can be non-blind or single-blind when used to measure bias for scanning and direct measurements.

- Matrix Spike Samples. Matrix spike samples are environmental samples that are spiked in the laboratory with a known concentration of a target analyte(s) to verify percent recoveries. They are primarily used to check sample matrix interferences but can also be used in the field to monitor instrument performance (EPA 1991g). Matrix Spike samples are often replicated to monitor a method's performance and evaluate bias and precision (when four or more pairs are analyzed). These replicates are often collectively referred to as a matrix spike/matrix spike duplicate (MS/MSD).

- Calibration Checks. Calibration checks are measurements performed to verify instrument performance each time an instrument is used (see Section 6.5.4). These checks may be qualitative or quantitative. Operators use qualitative checks to determine if an instrument is operating properly and can be used to perform measurements. Quantitative calibration checks require a specified protocol to measure a calibration source with a known instrument response, and the results are documented to provide a record of instrument precision and bias. The results of quantitative calibration checks are typically recorded on a control chart (see Section 6.2.2.7). Note that the calibration check source does not need to be traceable for qualitative or quantitative calibration checks as long as the instrument response has been adequately established (see Section 6.5.4). Because calibration checks are non-blind measurements they are only recommended when other types of QC measurements are not available.

Quality control measurements can also be used to estimate bias caused by contamination.

- Background Measurement. A background measurement is a measurement performed upgradient of the area of potential contamination (either onsite or offsite) where there is little or no chance of migration of the contaminants of concern (EPA 1991g). Background measurements are performed in the background reference area (Section 4.5), determine the natural composition and variability of the material of interest (especially important in areas with high concentrations of naturally occurring radionuclides), and are considered "clean." They provide a basis for comparison of contaminant concentration levels with measurements performed in the survey unit when the statistical tests described in Chapter 8 are performed.

- Measurement Blanks. Measurement blanks are samples prepared in the laboratory using certified clean sand or soil and brought to the field to monitor contamination for scanning and direct measurements. A measurement blank is used to evaluate contamination error associated with the instrument used to perform measurements in the field. Measurement blanks are recommended for determining bias resulting from contamination of instruments used for scanning and direct measurements.

6.2.2.3 Representativeness

Representativeness is a measure of the degree to which data accurately and precisely represent a characteristic of a population parameter at a sampling point (ASQC 1995) or measurement location. Representativeness is a qualitative term that is reflected in the survey design through the selection of a measurement method (*e.g.*, direct measurement or sampling).

Sample collection and analysis is typically less representative of true radionuclide concentrations at a specific measurement location than performing a direct measurement. This is caused by the additional steps required in collecting and analyzing samples, such as sample collection, field sample preparation, laboratory sample preparation, and radiochemical analysis. However, direct measurement techniques with acceptable detection limits are not always available. The location of the direct measurement is determined in Section 5.5.2.5, where random and systematic survey designs are selected based on survey unit classification. The coverage for a survey unit using scanning techniques is discussed in Section 5.5.3 and is also based primarily on survey unit classification. Because scanning locations are often selected based on professional judgment for survey units with less than 100% coverage, representativeness of these locations may be a concern. For both scanning and direct measurements the measurement locations and method for performing the measurements should be compared to the modeling assumptions used to develop the DCGLs.

6.2.2.4 Comparability

Comparability is a qualitative term that expresses the confidence that two data sets can contribute to a common analysis and interpolation. Generally, comparability is provided by using the same measurement system for all analyses of a specific radionuclide. Comparability is usually not an issue except in cases where historical data has been collected and is being compared to current analytical results, or when multiple laboratories are used to provide results as part of a single survey design.

6.2.2.5 Completeness

Completeness is a measure of the amount of valid data obtained from the measurement system. This is expressed as a percentage of the number of valid measurements that should have been

collected. Completeness is of greater concern for laboratory analyses than for direct measurements because the consequences of incomplete data often require the collection of additional data. Completeness is a concern for scanning only if the scanning results are invalidated for some reason. Direct measurements and scans can usually be repeated fairly easily while the personnel performing the measurements are still in the field. For this reason MARSSIM strongly recommends that scanning and direct measurement results be evaluated as soon as possible. Direct measurements performed on a systematic grid to locate areas of elevated activity are also a concern for completeness. If one direct measurement result is not valid, the entire survey design for locating areas of elevated activity may be invalidated.

6.2.2.6 Other Data Quality Indicators

Several additional data quality indicators that influence the final status survey design are identified as DQOs in Section 2.3.1. Many of these (*e.g.*, selection and classification of survey units, decision error rates, variability in the contaminant concentration, lower bound of the gray region) are used to determine the number of measurements and are discussed in detail in Section 5.5.2. The method detection limit is directly related to the selection of a measurement method and a specific measurement system.

Scanning and direct measurement techniques should be capable of measuring levels below the established DCGLs— detection limits of 10-50% of the DCGL should be the target (see Section 6.7). Cost, time, best available technology, or other constraints may create situations where the above stated sensitivities are deemed impractical. Under these circumstances, higher detection sensitivities may be acceptable. Although service providers and instrument manufacturers will state detection limits, these sensitivities are usually based on ideal or optimistic situations and may not be achievable under site-specific measurement conditions. Detection limits are subject to variation from measurement to measurement, instrument to instrument, operator to operator, and procedure to procedure. This variation depends on geometry, background, instrument calibration, abundance of the radiations being measured, counting time, operator training, operator experience, self-absorption in the medium being measured, and interferences from radionuclides or other materials present in the medium. The detection limit that is achievable in practice should not exceed the DCGL.

6.2.2.7 Using Control Charts to Provide Control of Field Measurement Systems

Control charts are commonly used in radioanalytical laboratories to monitor the performance of laboratory instruments. Control charts are also useful for monitoring the performance of field instruments and can be used to help control field measurement systems.

A control chart is a graphical plot of measurement results with respect to time or sequence of measurement, together with limits within in which the measurement values are expected to lie

when the system is in a state of statistical control (DOE 1995). Calibration check results are typically plotted on control charts for field measurements. However, control charts may be developed for any measurements where the expected performance is established and documented. A separate set of control charts for monitoring each type of measurement (*e.g.*, calibration check, background, measurement of PE samples) should be developed for each instrument.

The control chart is constructed by preparing a graph showing the arithmetic mean and the control limits as horizontal lines. The recommended control limits are two standard deviations above and below the mean, and three standard deviations above and below the mean. The measurement results in the appropriate units are shown on the y-axis and time or sequence is plotted using the x-axis. Detailed guidance on the development and use of control charts is available in *Quality Assurance of Chemical Measurements* (Taylor 1987) and *Statistical Methods for Quality Improvement* (Kume 1985).

As the quality control or other measurements are performed, the results are entered on the control chart. If the results are outside the control limits or show a particular trend or tendency, then the process is not in control. The control chart documents the performance of the measurement system during the time period of interest.

Quality control measurements for field instruments may be difficult or expensive to obtain for some surveys. In these cases control charts documenting instrument performance may represent the only determination of precision and bias for the survey. Because control charts are non-blind measurements they are generally not appropriate for estimating precision and bias. However, the control chart documents the performance of the field instruments. Provided the checks for precision and bias fall within the control limits, the results obtained using that instrument should be acceptable for the survey.

6.3 Selecting a Service Provider to Perform Field Data Collection Activities

One of the first steps in designing a survey is to select a service provider to perform field data collection activities. MARSSIM recommends that this selection take place early in the planning process so that the service provider can provide information during survey planning and participate in the design of the survey. Service providers may include in-house experts in field measurements and sample collection, health physics companies, or environmental engineering firms among others.

When the service provider is not part of the organization responsible for the site, these services are obtained using some form of procurement mechanism. Examples of procurement mechanisms include purchase orders or contracts. A graded approach should be used in determining the appropriate method for procuring services.

Potential service providers should be evaluated to determine their ability to perform the necessary analyses. For large or complex sites, this evaluation may take the form of a pre-award audit. The results of this audit provide a written record of the decision to use a specific service provider. For less complex sites or facilities, a review of the potential service provider's qualifications is sufficient for the evaluation.

There are six criteria that should be reviewed during this evaluation:

- Does the service provider possess the validated Standard Operating Procedures (SOPs), appropriate instrumentation, and trained personnel necessary to perform the field data collection activities? Field data collection activities (e.g., scanning surveys, direct measurements, and sample collection) are defined by the data needs identified by the DQO process.
- Is the service provider experienced in performing the same or similar data collection activities?
- Does the service provider have satisfactory performance evaluation or technical review results? The service provider should be able to provide a summary of QA audits and QC measurement results to demonstrate proficiency. Equipment calibrations should be performed using National Institute of Standards and Technology (NIST) traceable reference radionuclide standards whenever possible.
- Is there an adequate capacity to perform all field data collection activities within the desired timeframe? This criterion considers the number of trained personnel and quantity of calibrated equipment available to perform the specified tasks.
- Does the service provider conduct an internal quality control review of all generated data that is independent of the data generators?
- Are there adequate protocols for method performance documentation, sample tracking and security (if necessary), and documentation of results?

Potential service providers should have an active and fully documented quality system in place.[1] This system should enable compliance with the objectives determined by the DQO process in Section 2.3 and Appendix D (see EPA 1994c). The elements of a quality management system are discussed in Section 9.1 (ASQC 1995, EPA 1994f).

[1] The quality management system is typically documented in one or more documents such as a Quality Management Plan (QMP) or Quality Assurance Manual (QAM). A description of quality systems is included in Section 9.1.

6.4 Measurement Methods

Measurement methods used to generate field data can be classified into two categories commonly known as scanning surveys and direct measurements. The decision to use a measurement method as part of the survey design is determined by the survey objectives and the survey unit classification. Scanning is performed to identify areas of elevated activity that may not be detected by other measurement methods. Direct measurements are analogous to collecting and analyzing samples to determine the average activity in a survey unit. Section 5.5.3 discusses combining scans and direct measurements in an integrated survey design.

6.4.1 Direct Measurements

To conduct direct measurements of alpha, beta, and photon surface activity, instruments and techniques providing the required detection sensitivity are selected. The type of instrument and method of performing the direct measurement are selected as dictated by the type of potential contamination present, the measurement sensitivity requirements, and the objectives of the radiological survey. Direct measurements are taken by placing the instrument at the appropriate distance[2] above the surface, taking a discrete measurement for a pre-determined time interval (*e.g.*, 10 s, 60 s, *etc.*), and recording the reading. A one minute integrated count technique is a practical field survey procedure for most equipment and provides detection sensitivities that are below most DCGLs. However, longer or shorter integrating times may be warranted (see Section 6.7.1 for information dealing with the calculation of direct measurement detection sensitivities).

Direct measurements may be collected at random locations in the survey unit. Alternatively, direct measurements may be collected at systematic locations and supplement scanning surveys for the identification of small areas of elevated activity (see Section 5.5.2.5). Direct measurements may also be collected at locations identified by scanning surveys as part of an investigation to determine the source of the elevated instrument response. Professional judgment may also be used to identify location for direct measurements to further define the areal extent of contamination and to determine maximum radiation levels within an area, although these types of direct measurements are usually associated with preliminary surveys (*i.e.*, scoping, characterization, remedial action support). All direct measurement locations and results should be documented.

[2] Measurements at several distances may be needed. Near-surface or surface measurements provide the best indication of the size of the contaminated region and are useful for model implementation. Gamma measurements at 1 m provide a good estimate of potential direct external exposure.

If the equipment and methodology used for scanning is capable of providing data of the same quality required for direct measurement (*e.g.*, detection limit, location of measurements, ability to record and document results), then scanning may be used in place of direct measurements. Results should be documented for at least the number of locations required for the statistical tests. In addition, some direct measurement systems may be able to provide scanning data, provided they meet the objectives of the scanning survey.

The following sections briefly describe methods used to perform direct measurements in the field. The instruments used to perform these measurements are described in more detail in Section 6.5.3 and Appendix H.

6.4.1.1 Direct Measurements for Photon Emitting Radionuclides

There are a wide variety of instruments available for measuring photons in the field (see Appendix H) but all of them are used in essentially the same way. The detector is set up at a specified distance from the surface being measured and data are collected for a specified period of time. The distance from the surface to the detector is generally determined by the calibration of the instrument because photons do not interact appreciably with air. When measuring x-rays or low-energy gamma rays, the detector is often placed closer to the surface to increase the counting efficiency. The time required to perform a direct measurement may vary from very short (*e.g.*, 10 seconds) to very long (*e.g.*, several days or weeks) depending on the type of detector and the required detection limit. In general, the lower the required detection limit the longer the time required to perform the measurement. A collimator may be used in areas where activity from adjacent or nearby areas might interfere with the direct measurement. The collimator (usually lead, tungsten, or steel) shields the detector from extraneous photons but allows activity from a specified area of the surface to reach the detector.

Example:

The portable germanium detector, or *in situ* gamma spectrometer, can be used to estimate gamma-emitting radionuclide concentrations in the field. As with the laboratory-based germanium detector with multichannel analyzer, *in situ* gamma spectrometry can discriminate among various radionuclides on the basis of characteristic gamma and x-ray energies to provide a nuclide-specific measurement. A calibrated detector measures the fluence rate of primary photons at specific energies that are characteristic of a particular radionuclide (NRC 1995b). This fluence rate can then be converted to units of concentration. Under certain conditions the fluence rate may be converted directly to dose or risk for a direct comparison to the release criterion rather than to the $DCGL_W$. Although this conversion is generally made, the fluence rate should be considered the fundamental parameter for assessing the level of radiation at a specific location because it is a directly measurable physical quantity.

For outdoor measurements, where the contaminant is believed to be distributed within the surface soil, it may be appropriate to assume a uniform depth profile when converting the fluence rate to a concentration. At sites where the soil is plowed or overturned regularly, this assumption is quite realistic because of the effects of homogenization. At sites where the activity was initially deposited on the surface and has gradually penetrated deeper over time, the actual depth profile will have a higher activity at the surface and gradually diminish with depth. In this case, the assumption of a uniform depth profile will estimate a higher radionuclide concentration relative to the average concentration over that depth. In cases where there is an inverted depth profile (*i.e.*, low concentration at the surface that increase with depth), the assumption of a uniform depth profile will underestimate the average radionuclide concentration over that depth. For this reason, MARSSIM recommends that soil cores be collected to determine the actual depth profile for the site. These soil cores may be collected during the characterization or remedial action support survey to establish a depth profile for planning a final status survey. The cores may also be collected during the final status survey to verify the assumptions used to develop the fluence-to-concentration correction.

For indoor measurements, uncollimated *in situ* measurements can provide useful information on the low-level average activity across an entire room. The position of the measurement within the room is not critical if the radionuclide of interest is not present in the building materials. A measurement of peak count rate can be converted to fluence rate, which can in turn be related to the average surface activity. The absence of a discernible peak would mean that residual activity could not exceed a certain average level. However, this method will not easily locate small areas of elevated activity. For situations where the activity is not uniformly distributed on the surface, a series of collimated measurements using a systematic grid allows the operator to identify general areas of elevated contamination.

The NRC draft report *Measurement Methods for Radiological Surveys in Support of New Decommissioning Criteria* (NRC 1995b) provides a detailed description of the theory and implementation of *in situ* gamma spectrometry. *In situ* spectrometry is provided as one example of a useful tool for performing direct measurements for particular scenarios, but interpretation of the instrument output in terms of radionuclide distributions is dependent on the assumptions used to calibrate the method site-specifically. The depth of treatment of this technique in this example is not meant to imply that *in situ* gamma spectrometry is preferred *a priori* over other appropriate measurement techniques described in this manual.

6.4.1.2 Direct Measurements for Alpha Emitting Radionuclides

Direct measurements for alpha-emitting radionuclides are generally performed by placing the detector on or near the surface to be measured. The limited range of alpha particles (*e.g.*, about 1 cm or 0.4 in. in air, less in denser material) means that these measurements are generally restricted to relatively smooth, impermeable surfaces such as concrete, metal, or drywall where the activity is present as surface contamination. In most cases, direct measurements of porous (*e.g.*, wood) and volumetric (*e.g.*, soil, water) material cannot meet the objectives of the survey. However, special instruments such as the long range alpha detector (see Appendix H) have been developed to measure the concentration of alpha emitting radionuclides in soil under certain conditions. Because the detector is used in close proximity to the potentially contaminated surface, contamination of the detector or damage to the detector caused by irregular surfaces need to be considered before performing direct measurements for alpha emitters.

6.4.1.3 Direct Measurements for Beta Emitting Radionuclides

Direct measurements for beta emitting radionuclides are generally performed by placing the detector on or near the surface to be measured, similar to measurements for alpha emitting radionuclides. These measurements are typically restricted to relatively smooth, impermeable surfaces where the activity is present as surface contamination. In most cases, direct measurements of porous (*e.g.*, wood) and volumetric (*e.g.*, soil, water) material cannot meet the objectives of the survey. However, special instruments such as large area gas-flow proportional counters (see Appendix H) and arrays of beta scintillators have been developed to measure the concentration of beta emitting radionuclides in soil under certain conditions. Similar to direct measurements for alpha emitting radionuclides, contamination of the detector and damage to the detector need to be considered before performing direct measurements for beta emitters.

6.4.2 Scanning Surveys

Scanning is the process by which the operator uses portable radiation detection instruments to detect the presence of radionuclides on a specific surface (*i.e.*, ground, wall, floor, equipment). The term scanning survey is used to describe the process of moving portable radiation detectors across a suspect surface with the intent of locating radionuclide contamination. Investigation levels for scanning surveys are determined during survey planning to identify areas of elevated activity. Scanning surveys are performed to locate radiation anomalies indicating residual gross activity that may require further investigation or action. These investigation levels may be based on the $DCGL_W$, the $DCGL_{EMC}$, or some other level as discussed in Section 5.5.2.6.

Small areas of elevated activity typically represent a small portion of the site or survey unit. Thus, random or systematic direct measurements or sampling on the commonly used grid spacing may have a low probability of identifying such small areas. Scanning surveys are often relatively quick and inexpensive to perform. For these reasons, scanning surveys are typically performed before direct measurements or sampling. This way time is not spent fully evaluating an area that may quickly prove to be contaminated above the investigation level during the scanning process. Scans are conducted which would be indicative of all radionuclides potentially present, based on the Historical Site Assessment, surfaces to be surveyed, and survey design objectives. Surrogate measurements may be utilized where appropriate (see Section 4.3.2). Documenting scanning results and observations from the field is very important. For example, a scan that identified relatively sharp increases in instrument response or identified the boundary of an area of increased instrument response should be documented. This information is useful when interpreting survey results.

The following sections briefly describe techniques used to perform scanning surveys for different types of radiation. The instruments used to perform these measurements are described in more detail in Section 6.5.3 and Appendix H.

6.4.2.1 Scanning for Photon Emitting Radionuclides

Sodium iodide survey meters (NaI(Tl) detectors) are normally used for scanning areas for gamma emitters because they are very sensitive to gamma radiation, easily portable and relatively inexpensive. The detector is held close to the ground surface (~6 cm or 2.5 in.) and moved in a serpentine (*i.e.*, snake like, "S" shaped) pattern while walking at a speed that allows the investigator to detect the desired investigation level. A scan rate of approximately 0.5 m/s is typically used for distributed gamma emitting contaminants in soil; however, this rate must be adjusted depending on the expected detector response and the desired investigation level. Discussion of scanning rates versus detection sensitivity for gamma emitters is provided in Section 6.7.2.1.

Sodium iodide survey meters are also used for scanning to detect areas with elevated areas of low-energy gamma and x-ray emitting radionuclides such as ^{241}Am and ^{239}Pu. Specially designed detectors, such as the FIDLER (field instrument for the detection of low energy radiation) probe with survey meter, are typically used to detect these types of radionuclides.

6.4.2.2 Scanning for Alpha Emitting Radionuclides

Alpha scintillation survey meters and thin window gas-flow proportional counters are typically used for performing alpha surveys. Alpha radiation has a very limited range and, therefore, instrumentation must be kept close to the surface—usually less than 1 cm (0.4 in.). For this reason, alpha scans are generally performed on relatively smooth, impermeable surfaces (*e.g.*,

concrete, metal, drywall) and not on porous material (*e.g.*, wood) or for volumetric contamination (*e.g.*, soil, water). In most cases, porous and volumetric contamination cannot be detected by scanning for alpha activity and meet the objectives of the survey because of high detection sensitivities. Under these circumstances, samples of the material are usually collected and analyzed as discussed in Chapter 7. Determining scan rates when surveying for alpha emitters is discussed in Section 6.7.2.2 and Appendix J.

6.4.2.3 Scanning for Beta Emitting Radionuclides

Thin window gas-flow proportional counters are normally used when surveying for beta emitters, although solid scintillators designed for this purpose are also available. Typically, the beta detector is held less than 2 cm from the surface and moved at a rate such that the desired investigation level can be detected. Low-energy (<100 keV) beta emitters are subject to the same interferences and self-absorption problems found with alpha emitting radionuclides, and scans for these radionuclides are performed under similar circumstances. Determination of scan rates when surveying for beta emitters is discussed in Section 6.7.2.1.

6.5 Radiation Detection Instrumentation

Traditional radiation instruments consist of two components: 1) a radiation detector, and 2) electronic equipment to provide power to the detector and to display or record radiation events. This section identifies and very briefly describes the types of radiation detectors and associated display or recording equipment that are applicable to survey activities in support of environmental assessment or remedial action. Each survey usually requires performing direct field measurements using portable instrumentation and collection of samples for laboratory analysis. The selection and proper use of appropriate instruments for both direct measurements and laboratory analyses will likely be the most critical factors in assuring that the survey accurately determines the radiological status of a site and meets the survey objectives. Chapter 7 provides specific information on laboratory analysis of collected samples. Appendix H contains instrument specific information for various types of field survey and laboratory analysis equipment currently in use.

6.5.1 Radiation Detectors

The particular capabilities of a radiation detector will establish its potential applications in conducting a specific type of survey. Radiation detectors can be divided into four general classes based on the detector material or the application. These categories are: 1) gas-filled detectors, 2) scintillation detectors, 3) solid-state detectors, and 4) passive integrating detectors.

6.5.1.1 Gas-Filled Detectors

Radiation interacts with the fill gas, producing ion-pairs that are collected by charged electrodes. Commonly used gas-filled detectors are categorized as ionization, proportional, or Geiger-Mueller (GM), referring to the region of gas amplification in which they are operated. The fill gas varies, but the most common are: 1) air, 2) argon with a small amount of organic methane (usually 10% methane by mass, referred to as P-10 gas), and 3) argon or helium with a small amount of a halogen such as chlorine or bromine added as a quenching agent.

6.5.1.2 Scintillation Detectors

Radiation interacts with a solid or liquid medium causing electronic transitions to excited states in a luminescent material. The excited states decay rapidly, emitting photons that in turn are captured by a photomultiplier tube. The ensuing electrical signal is proportional to the scintillator light output, which, under the right conditions, is proportional to the energy loss that produced the scintillation. The most common scintillant materials are NaI(Tl), ZnS(Ag), Cd(Te), and CsI(Tl) which are used in traditional radiation survey instruments such as the NaI(Tl) detector used for gamma surveys and the ZnS(Ag) detector for alpha surveys.

6.5.1.3 Solid-State Detectors

Radiation interacting with a semiconductor material creates electron-hole pairs that are collected by a charged electrode. The design and operating conditions of a specific solid-state detector determines the types of radiations (alpha, beta, and/or gamma) that can be measured, the detection level of the measurements, and the ability of the detector to resolve the energies of the interacting radiations. The semiconductor materials currently being used are germanium and silicon which are available in both n and p types in various configurations.

Spectrometric techniques using these detectors provide a marked increase in sensitivity in many situations. When a particular radionuclide contributes only a fraction of the total particle or photon fluence, or both, from all sources (natural or manmade background), gross measurements are inadequate and nuclide-specific measurements are necessary. Spectrometry provides the means to discriminate among various radionuclides on the basis of characteristic energies. *In-situ* gamma spectrometry is particularly effective in field measurements since the penetrating nature of the radiation allows one to "see" beyond immediate surface contamination. The availability of large, high efficiency germanium detectors permits measurement of low abundance gamma emitters such as ^{238}U as well as low energy emitters such as ^{241}Am and ^{239}Pu.

6.5.1.4 Passive Integrating Detectors

There is an additional class of instruments that consists of passive, integrating detectors and associated reading/analyzing instruments. The integrated ionization is read using a laboratory or hand-held reader. This class includes thermoluminescence dosimeters (TLDs) and electret ion chambers (EICs). Because these detectors are passive and can be exposed for relatively long periods of time, they can provide better sensitivity for measuring low activity levels such as free release limits or for continuing surveillance. The ability to read and present data onsite is a useful feature and such systems are comparable to direct reading instruments.

The scintillation materials in Section 6.5.1.2 are selected for their prompt fluorescence characteristics. In another class of inorganic crystals, called TLDs, the crystal material and impurities are chosen so that the free electrons and holes created following the absorption of energy from the radiation are trapped by impurities in the crystalline lattice thus locking the excitation energy in the crystal. Such materials are used as passive, integrating detectors. After removal from the exposure area, the TLDs are heated in a reader which measures the total amount of light produced when the energy is released. The total amount of light is proportional to the number of trapped, excited electrons, which in turn is proportional to the amount of energy absorbed from the radiation. The intensity of the light emitted from the thermoluminescent crystals is thus directly proportional to the radiation dose. TLDs come in a large number of materials, the most common of which are LiF, CaF_2:Mn, CaF_2:Dy, $CaSO_4$:Mn, $CaSO_4$:Dy, Al_2O_3:C.

The electret ion chamber consists of a very stable electret (a charged Teflon® disk) mounted inside a small chamber made of electrically charged plastic. The ions produced inside this air filled chamber are collected onto the electret, causing a reduction of its surface charge. The reduction in charge is a function of the total ionization during a specific monitoring period and the specific chamber volume. This change in voltage is measured with a surface potential voltmeter.

6.5.2 Display and Recording Equipment

Radiation detectors are connected to electronic devices to 1) provide a source of power for detector operation, and 2) enable measurement of the quantity and/or quality of the radiation interactions that are occurring in the detector. The quality of the radiation interaction refers to the amount of energy transferred to the detector. In many cases, radiation interacts with other material (*e.g.*, air) prior to interacting with the detector, or only partially interacts with the detector (*e.g.*, Compton scattering for photons). Because the energy recorded by the detector is affected, there is an increased probability of incorrectly identifying the radionuclide.

The most common recording or display device used for portable radiation measurement systems is a ratemeter. This device provides a display on an analog meter representing the number of events occurring over some time period (*e.g.*, counts per minute). Digital ratemeters are also commercially available. The number of events can also be accumulated over a preset time period using a digital scaling device. The resulting information from a scaling device is the total number of events that occurred over a fixed period of time, where a ratemeter display varies with time and represents a short term average of the event rate. Determining the average level on a ratemeter will require judgment by the user, especially when a low frequency of events results in significant variations in the meter reading.

Pulse height analyzers are specialized electronic devices designed to measure and record the number of pulses or events that occur at different pulse height levels. These types of devices are used with detectors which produce output pulses that are proportional in height to the energy deposited within them by the interacting radiation. They can be used to record only those events occurring in a detector within a single band of energy or can simultaneously record the events in multiple energy ranges. In the former case, the equipment is known as a single-channel analyzer; the latter application is referred to as a multichannel analyzer.

6.5.3 Instrument Selection

Radiation survey parameters that might be needed for site release purposes include surface activities, exposure rates, and radionuclide concentrations in soil. To determine these parameters, field measurements and laboratory analyses may be necessary. For certain radionuclides or radionuclide mixtures, both alpha and beta radiations may have to be measured. In addition to assessing average radiological conditions, the survey objectives should address identifying small areas of elevated activity and determining the extent and level of residual radioactivity.

Additionally, the potential uses of radiation instruments can vary significantly depending on the specific design and operating criteria of a given detector type. For example, a NaI(Tl) scintillator can be designed to be very thin with a low atomic number entrance window (*e.g.*, beryllium) such that the effective detection capability for low energy photons is optimized. Conversely, the same scintillant material can be fabricated as a thick cylinder in order to optimize the detection probability for higher energy photons. On the recording end of a detection system, the output could be a ratemeter, scaler, or multichannel analyzer as described in Section 6.5.2. Operator variables such as training and level of experience with specific instruments should also be considered.

With so many variables, it is highly unlikely that any single instrument (detector and readout combination) will be capable of adequately measuring all of the radiological parameters necessary to demonstrate that criteria for release have been satisfied. It is usually necessary to select multiple instruments to perform the variety of measurements required.

Selection of instruments will require an evaluation of a number of situations and conditions. Instruments must be stable and reliable under the environmental and physical conditions where they will be used, and their physical characteristics (size and weight) should be compatible with the intended application. The instrument must be able to detect the type of radiation of interest, and the measurement system should be capable of measuring levels that are less than the DCGL (see Section 6.7).

For gamma radiation scanning, a scintillation detector/ratemeter combination is the usual instrument of choice. A large-area proportional detector with a ratemeter is recommended for scanning for alpha and beta radiations where surface conditions and locations permit; otherwise, an alpha scintillation or thin-window GM detector (for beta surveys) may be used.

For direct gamma measurements, a pressurized ionization chamber or *in-situ* gamma spectroscopy system is recommended. As an option, a NaI(Tl) scintillation detector may be used if cross-calibrated to a pressurized ion chamber or calibrated for the specific energy of interest. The same alpha and beta detectors identified above for scanning surveys are also recommended for use in direct measurements.

There are certain radionuclides that, because of the types, energies, and abundances of their radiations, will be essentially impossible to measure at the guideline levels, under field conditions, using state-of-the-art instrumentation and techniques. Examples of such radionuclides include very low energy pure beta emitters, such as ^{3}H and ^{63}Ni, and low energy photon emitters, such as ^{55}Fe and ^{125}I. Pure alpha emitters dispersed in soil or covered with some absorbing layer will not be detectable because the alpha radiation will not penetrate through the media or covering to reach the detector. A common example of such a condition would be ^{230}Th surface contamination covered by paint, dust, oil, or moisture. In such circumstances, sampling and laboratory analysis would be required to measure the residual activity levels unless surrogate radionuclides are present as discussed in Section 4.3.2.

The number of possible design and operating schemes for each of the different types of detectors is too large to discuss in detail within the context of this document. For a general overview, lists of common radiation detectors along with their usual applications during surveys are provided in Tables 6.1 through 6.3. Appendix H contains specific information for various types of field survey and laboratory analysis equipment currently in use. Continual development of new technologies will result in changes to these listings.

Table 6.1 Radiation Detectors with Applications to Alpha Surveys

Detector Type	Detector Description	Application	Remarks
Gas Proportional	<1 mg/cm^2 window; probe area 50 to 1000 cm^2 <0.1 mg/cm^2 window; probe area 10 to 20 cm^2 No window (internal proportional)	Surface scanning; surface contamination measurement Laboratory measurement of water, air, and smear samples Laboratory measurement of water, air, and smear samples	Requires a supply of appropriate fill gas
Air Proportional	<1 mg/cm^2 window; probe area ~50 cm^2	Useful in low humidity conditions	
Scintillation	ZnS(Ag) scintillator; probe area 50 to 100 cm^2 ZnS(Ag) scintillator; probe area 10 to 20 cm^2 Liquid scintillation cocktail containing sample	Surface contamination measurements, smears Laboratory measurement of water, air, and smear samples Laboratory analysis, spectrometry capabilities	
Solid State	Silicon surface barrier detector	Laboratory analysis by alpha spectrometry	
Passive, integrating electret ion chamber	<0.8 mg/cm^2 window, also window-less, window area 50-180 cm^2, chamber volume 50-1,000 ml	Contamination on surfaces, in pipes and in soils	Useable in high humidity and temperature

6.5.4 Instrument Calibration

Calibration refers to the determination and adjustment of the instrument response in a particular radiation field of known intensity. Proper calibration procedures are an essential requisite toward providing confidence in measurements made to demonstrate compliance with cleanup criteria. Certain factors, such as energy dependence and environmental conditions, require consideration in the calibration process, depending on the conditions of use of the instrument in the field. Routine calibration of radiation detection instruments refers to calibration for normal use under typical field conditions. Considerations for the use and calibration of instruments include:

Table 6.2 Radiation Detectors with Applications to Beta Surveys

Detector Type	Detector Description	Application	Remarks
Gas Proportional	<1 mg/cm^2 window; probe area 50 to 1,000 cm^2	Surface scanning; surface contamination measurement	Requires a supply of appropriate fill gas
	<0.1 mg/cm^2 window; probe area 10 to 20 cm^2	Laboratory measurement of water, air, smear, and other samples	
	No window (internal proportional)	Laboratory measurement of water, air, smear, and other samples	Can be used for measuring very low-energy betas
Ionization (non-pressurized)	1-7 mg/cm^2 window	Contamination measurements; skin dose rate estimates	
Geiger-Mueller	<2 mg/cm^2 window; probe area 10 to 100 cm^2	Surface scanning; contamination measurements; laboratory analyses	
	Various window thickness; few cm^2 probe face	Special scanning applications	
Scintillation	Liquid scintillation cocktail containing sample	Laboratory analysis; spectrometry capabilities	
	Plastic scintillator	Contamination measurements	
Passive, integrating electret ion chamber	7 mg/cm^2 window, also window-less, window area 50-180 cm^2, chamber volume 50-1,000 ml	Low energy beta including H-3 contamination on surfaces and in pipes	Useable in high humidity and temperature

- use of the instrument for radiation of the type for which the instrument is designed
- use of the instrument for radiation energies within the range of energies for which the instrument is designed
- use under environmental conditions for which the instrument is designed
- use under influencing factors, such as magnetic and electrostatic fields, for which the instrument is designed
- use of the instrument in an orientation such that geotropic effects are not a concern
- use of the instrument in a manner that will not subject the instrument to mechanical or thermal stress beyond that for which it is designed

Table 6.3 Radiation Detectors with Applications to Gamma Surveys

Detector Type	Detector Description	Application	Remarks
Gas Ionization	Pressurized ionization chamber; Non-pressurized ionization chamber	Exposure rate measurements	
Geiger-Mueller	Pancake (<2 mg/cm^2 window) or side window (~30 mg/cm^2)	Surface scanning; exposure rate correlation (side window in closed position)	Low relative sensitivity to gamma radiation
Scintillation	NaI(Tl) scintillator; up to 5 cm by 5 cm	Surface scanning; exposure rate correlation	High sensitivity; Cross calibrate with PIC (or equivalent) or for specific site gamma energy mixture for exposure rate measurements.
	NaI(Tl) scintillator; large volume and "well" configurations	Laboratory gamma spectrometry	
	CsI or NaI(Tl) scintillator; thin crystal	Scanning; low-energy gamma and x-rays	Detection of low-energy radiation
	Organic tissue equivalent (plastics)	Dose equivalent rate measurements	
Solid State	Germanium semi-conductor	Laboratory and field gamma spectrometry and spectroscopy	
Passive, integrating electret ion chamber	7 mg/cm^2 window, also window-less, window area 50-180 cm^2, chamber volume 50-1,000 ml		Useable in high humidity and temperature

Routine calibration commonly involves the use of one or more sources of a specific radiation type and energy, and of sufficient activity to provide adequate field intensities for calibration on all ranges of concern.

Actual field conditions under which the radiation detection instrument will be used may differ significantly from those present during routine calibration. Factors which may affect calibration validity include:

- the energies of radioactive sources used for routine calibration may differ significantly from those of radionuclides in the field
- the source-detector geometry (*e.g.*, point source or large area distributed source) used for routine calibration may be different than that found in the field
- the source-to-detector distance typically used for routine calibration may not always be achievable in the field
- the condition and composition of the surface being monitored (*e.g.*, sealed concrete, scabbled concrete, carbon steel, stainless steel, and wood) and the presence of overlaying material (*e.g.*, water, dust, oil, paint) may result in a decreased instrument response relative to that observed during routine calibration

If the actual field conditions differ significantly from the calibration assumptions, a special calibration for specific field conditions may be required. Such an extensive calibration need only be done once to determine the effects of the range of field conditions that may be encountered at the site. If responses under routine calibration conditions and proposed use conditions are significantly different, a correction factor or chart should be supplied with the instrument for use under the proposed conditions.

As a minimum, each measurement system (detector/readout combination) should be calibrated annually and response checked with a source following calibration (ANSI 1996). Instruments may require more frequent calibration if recommended by the manufacturer. Re-calibration of field instruments is also required if an instrument fails a performance check or if it has undergone repair or any modification that could affect its response.

The user may decide to perform calibrations following industry recognized procedures (ANSI 1996b, DOE Order 5484.1, NCRP 1978, NCRP 1985, NCRP 1991, ISO 1988, HPS 1994a, HPS 1994b), or the user can choose to obtain calibration by an outside service, such as a major instrument manufacturer or a health physics services organization.

Calibration sources should be traceable to the National Institute of Standards and Technology (NIST). Where NIST traceable standards are not available, standards obtained from an industry recognized organization (*e.g.*, the New Brunswick Laboratory for various uranium standards) may be used.

Calibration of instruments for measurement of surface contamination should be performed such that a direct instrument response can be accurately converted to the 4π (total) emission rate from the source. An accurate determination of activity from a measurement of count rate above a surface in most cases is an extremely complex task because of the need to determine appropriate chacteristics of the source including decay scheme, geometry, energy, scatter, and self-absorption. For the purpose of release of contaminated areas from radiological control, measurements must provide sufficient accuracy to ensure that cleanup standards have been achieved. Inaccuracies in

measurements should be controlled in a manner that minimizes the consequences of decision errors. The variables that affect instrument response should be understood well enough to ensure that the consequences of decision errors are minimized. Therefore, the calibration should account for the following factors (where necessary):

- Calibrations for point and large area source geometries may differ, and both may be necessary if areas of activity smaller than the probe area and regions of activity larger than the probe area are present.
- Calibration should either be performed with the radionuclide of concern, or with appropriate correction factors developed for the radionuclide(s) present based on calibrations with nuclides emitting radiations similar to the radionuclide of concern.
- For portable instrumentation, calibrations should account for the substrate of concern (*i.e.*, concrete, steel) or appropriate correction factors developed for the substrates relative to the actual calibration standard substrate. This is especially important for beta emitters because backscatter is significant and varies with the composition of the substrate. Conversion factors developed during the calibration process should be for the same counting geometry to be used during the actual use of the detector.

For cleanup standards for building surfaces, the contamination level is typically expressed in terms of the particle emission rate per unit time per unit area, normally Bq/m^2 or disintegrations per minute (dpm) per 100 cm^2. In many facilities, surface contamination is assessed by converting the instrument response (in counts per minute) to surface activity using one overall total efficiency. The total efficiency may be considered to represent the product of two factors, the instrument (detector) efficiency, and the source efficiency. Use of the total efficiency is not a problem provided that the calibration source exhibits characteristics similar to the surface contamination (*i.e.*, radiation energy, backscatter effects, source geometry, self-absorption). In practice, this is hardly the case; more likely, instrument efficiencies are determined with a clean, stainless steel source, and then those efficiencies are used to determine the level of contamination on a dust-covered concrete surface. By separating the efficiency into two components, the surveyor has a greater ability to consider the actual characteristics of the surface contamination.

The instrument efficiency is defined as the ratio of the net count rate of the instrument and the surface emission rate of a source for a specified geometry. The surface emission rate is defined as the number of particles of a given type above a given energy emerging from the front face of the source per unit time. The surface emission rate is the 2π particle fluence that embodies both the absorption and scattering processes that effect the radiation emitted from the source. Thus, the instrument efficiency is determined by the ratio of the net count rate and the surface emission rate.

The instrument efficiency is determined during calibration by obtaining a static count with the detector over a calibration source that has a traceable activity or surface emission rate. In many cases, a source emission rate is measured by the manufacturer and certified as NIST traceable. The source activity is then calculated from the surface emission rate based on assumed backscatter and self-absorption properties of the source. The maximum value of instrument efficiency is 1.

The source efficiency is defined as the ratio of the number of particles of a given type emerging from the front face of a source and the number of particles of the same type created or released within the source per unit time. The source efficiency takes into account the increased particle emission due to backscatter effects, as well as the decreased particle emission due to self-absorption losses. For an ideal source (*i.e.*, no backscatter or self-absorption), the value of the source efficiency is 0.5. Many real sources will exhibit values less than 0.5, although values greater than 0.5 are possible, depending on the relative importance of the absorption and backscatter processes.

Source efficiencies may be determined experimentally. Alternatively, ISO-7503-1 (ISO 1988) makes recommendations for default source efficiencies. A source efficiency of 0.5 is recommended for beta emitters with maximum energies above 0.4 MeV. Alpha emitters and beta emitters with maximum beta energies between 0.15 and 0.4 MeV have a recommended source efficiency of 0.25. Source efficiencies for some common surface materials and overlaying material are provided in NUREG-1507 (NRC 1997b).

Instrument efficiency may be affected by detector-related factors such as detector size (probe surface area), window density thickness, geotropism, instrument response time, counting time (in static mode), scan rate (in scan mode), and ambient conditions such as temperature, pressure, and humidity. Instrument efficiency also depends on solid angle effects, which include source-to-detector distance and source geometry.

Source efficiency may be affected by source-related factors such as the type of radiation and its energy, source uniformity, surface roughness and coverings, and surface composition (*e.g.*, wood, metal, concrete).

The calibration of gamma detectors for the measurement of photon radiation fields should also provide reasonable assurance of acceptable accuracy in field measurements. Use of these instruments for demonstration of compliance with cleanup standards is complicated by the fact that most cleanup levels produce exposure rates of at most a few μR/h. Several of the portable survey instruments currently available in the United States for exposure rate measurements of ~1 μR/h (often referred to as micro-R meters) have full scale intensities of ~3 to 5 μR/h on the first range. This is below the ambient background for most low radiation areas and most calibration laboratories. (A typical background dose equivalent rate of 100 mrem/y gives a

background exposure rate of about 10 μR/h.) Even on the second range, the ambient background in the calibration laboratory is normally a significant part of the range and must be taken into consideration during calibration. The instruments commonly are not energy-compensated and are very sensitive to the scattered radiation that may be produced by the walls and floor of the room or additional shielding required to lower the ambient background.

Low intensity sources and large distances between the source and detector can be used for low-level calibrations if the appropriate precautions are taken. Field characterization of low-level sources with traceable transfer standards is difficult because of the poor signal-to-noise ratio in the standard chamber. In order to achieve adequate ionization current, the distance between the standard chamber and the source generally will be as small as possible while still maintaining good geometry (5 to 7 detector diameters). Generally it is not possible to use a standard ionization chamber to characterize the field at the distance necessary to reduce the field to the level required for calibration. A high quality GM detector, calibrated as a transfer standard, may be useful at low levels.

Corrections for scatter can be made using a shadow-shield technique in which a shield of sufficient density and thickness to eliminate virtually all the primary radiation is placed about midway between the source and the detector. The dimensions of the shield should be the minimum required to reduce the primary radiation intensity at the detector location to less than 2% of its unshielded value. The change in reading caused by the shield being removed is attributed to the primary field from the source at the detector position.

In some instruments that produce pulses (GM counters or scintillation counters), the detector can be separated electronically from the readout electronics and the detector output can be simulated with a suitable pulser. Caution must be exercised to ensure that either the high voltage is properly blocked or that the pulser is designed for this application. If this can be accomplished, the instrument can first be calibrated on a higher range that is not affected by the ambient background and in a geometry where scatter is not a problem and, after disconnecting the detector, to provide the pulse-rate from the pulser which will give the same instrument response. The pulse rate can then be related to field strength and reduced to give readings on lower ranges (with the detector disconnected) even below the ambient background. This technique does not take account of any inherent detector background independent of the external background.

Ionization chambers are commonly used to measure radiation fields at very low levels. In order to obtain the sensitivity necessary to measure these radiation levels, the instruments are frequently very large and often pressurized. These instruments have the same calibration problems as the more portable micro-R meters described above. The same precautions (shadow shield) must be taken to separate the response of the instrument to the source and to scattered radiation. Generally, it is not possible to substitute an electronic pulser for the radiation field in these instruments.

For energy-dependent gamma scintillation instruments, such as NaI(Tl) detectors, calibration for the gamma energy spectrum at a specific site may be accomplished by comparing the instrument response to that of a pressurized ionization chamber, or equivalent detector, at different locations on the site. Multiple radionuclides with various photon energies may also be used to calibrate the system for the specific energy of interest.

In the interval between calibrations, the instrument should receive a performance check prior to use. In some cases, a performance check following use may also provide valuable information. This calibration check is merely intended to establish whether or not the instrument is operating within certain specified, rather large, uncertainty limits. The initial performance check should be conducted following the calibration by placing the source in a fixed, reproducible location and recording the instrument reading. The source should be identified along with the instrument, and the same check source should be used in the same fashion to demonstrate the instrument's operability on a daily basis when the instrument is in use. For analog readout (count rate) instruments, a variation of ± 20% is usually considered acceptable. Optionally, instruments that integrate events and display the total on a digital readout typically provide an acceptable average response range of 2 or 3 standard deviations. This is achieved by performing a series of repetitive measurements (10 or more is suggested) of background and check source response and determining the average and standard deviation of those measurements. From a practical standpoint, a maximum deviation of ± 20% is usually adequate when compared with other uncertainties associated with the use of the equipment. The amount of uncertainty allowed in the response checks should be consistent with the level of uncertainty allowed in the final data. Ultimately the decision maker determines what level of uncertainty is acceptable.

Instrument response, including both the background and check source response of the instrument, should be tested and recorded at a frequency that ensures the data collected with the equipment is reliable. For most portable radiation survey equipment, MARSSIM recommends that a response check be performed twice daily when in use—typically prior to beginning the day's measurements and again following the conclusion of measurements on that same day. Additional checks can be performed if warranted by the instrument and the conditions under which it is used. If the instrument response does not fall within the established range, the instrument is removed from use until the reason for the deviation can be resolved and acceptable response again demonstrated. If the instrument fails the post-survey source check, all data collected during that time period with the instrument must be carefully reviewed and possibly adjusted or discarded, depending on the cause of the failure. Ultimately, the frequency of response checks must be balanced with the stability of the equipment being used under field conditions and the quantity of data being collected. For example, if the instrument experiences a sudden failure during the course of the day's work due to physical harm, such as a punctured probe, then the data collected up until that point is probably acceptable even though a post-use performance check cannot be performed. Likewise, if no obvious failure occurred but the instrument failed the post-use response check, then the data collected with that instrument since the last response check should be viewed with

great skepticism and possibly re-collected or randomly checked with a different instrument. Additional corrective action alternatives are presented in Section 9.3. If re-calibration is necessary, acceptable response ranges must be reestablished and documented.

Record requirements vary considerably and depend heavily on the needs of the user. While Federal and State regulatory agencies all specify requirements, the following records should be considered a minimum.

Laboratory Quality Control

- records documenting the traceabililty of radiological standards
- records documenting the traceability of electronic test equipment

Records for Instruments to be Calibrated

- date received in the calibration laboratory
- initial condition of the instrument, including mechanical condition (*e.g.*, loose or broken parts, dents, punctures), electrical condition (*e.g.*, switches, meter movement, batteries), and radiological condition (presence or absence of contamination)
- calibrator's records including training records and signature on calibration records
- calibration data including model and serial number of instrument, date of calibration, recommended recalibration date, identification of source(s) used, "as found" calibration results, and final calibration results—"as returned" for use.

In addition, records of instrument problems, failures, and maintenance can be included and are useful in assessing performance and identifying possible needs for altered calibration frequencies for some instruments. Calibration records should be maintained at the facility where the instruments are used as permanent records, and should be available either as hard copies or in safe computer storage.

6.6 Data Conversion

This section describes methods for converting survey data to appropriate units for comparison to radiological criteria. As stated in Chapter 4, conditions applicable to satisfying decommissioning requirements include determining that any residual contamination will not result in individuals being exposed to unacceptable levels of radiation and/or radioactive materials.

Radiation survey data are usually obtained in units, such as the number of counts per unit time, that have no intrinsic meaning relative to DCGLs. For comparison of survey data to DCGLs, the survey data from field and laboratory measurements should be converted to DCGL units.

6.6.1 Surface Activity

When measuring surface activity, it is important to account for the physical surface area assessed by the detector in order to make probe area corrections and report data in the proper units (*i.e.*, Bq/m^2, dpm/100 cm^2). This is termed the *physical probe area*. A common misuse is to make probe area corrections using the *effective probe area* which accounts for the amount of the physical probe area covered by a protective screen. Figure 6.1 illustrates the difference between the physical probe area and the effective probe area. The physical probe area is used because the reduced detector response due to the screen is accounted for during instrument calibration.

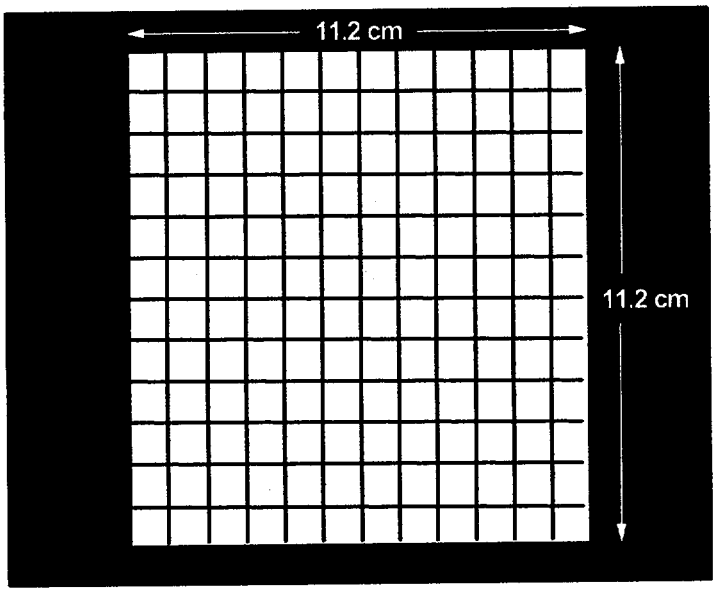

Figure 6.1 The Physical Probe Area of a Detector

The conversion of instrument display in counts to surface activity units is obtained using the following equation.

$$Bq/m^2 = \frac{\frac{C_s}{T_s}}{(\varepsilon_T \times A)} \qquad (6\text{-}1)$$

where

C_s	=	integrated counts recorded by the instrument
T_s	=	time period over which the counts were recorded in seconds
ε_T	=	total efficiency of the instrument in counts per disintegration, effectively the product of the instrument efficiency (ε_i) and the source efficiency (ε_s)
A	=	physical probe area in m^2

To convert instrument counts to conventional surface activity units, Equation 6-1 can be modified as shown in Equation 6-2.

$$\frac{dpm}{100\ cm^2} = \frac{\frac{C_s}{T_s}}{(\varepsilon_T) \times (A/100)} \qquad (6\text{-}2)$$

where T_s is recorded in minutes instead of seconds, and A is recorded in cm^2 instead of m^2.

Some instruments have background counts associated with the operation of the instrument. A correction for instrument background can be included in the data conversion calculation as shown in Equation 6-3. Note that the instrument background is not the same as the measurements in the background reference area used to perform the statistical tests described in Chapter 8.

$$Bq/m^2 = \frac{\frac{C_s}{T_s} - \frac{C_b}{T_b}}{(\varepsilon_T \times A)} \qquad (6\text{-}3)$$

where

C_b	=	background counts recorded by the instrument
T_b	=	time period over which the background counts were recorded in seconds

Equation 6-3 can be modified to provide conventional surface activity units as shown in Equation 6-4.

$$\frac{dpm}{100\ cm^2} = \frac{\frac{C_s}{T_s} - \frac{C_b}{T_b}}{(\text{å}_T) \times (A/100)} \qquad (6\text{-}4)$$

where T_s and T_b are recorded in minutes instead of seconds and A is recorded in cm^2 instead of m^2.

The presence of multiple radionuclides at a site requires additional considerations for demonstrating compliance with a dose- or risk-based regulation. As demonstrated in Section 4.3.2, a gross activity DCGL should be determined. For example, consider a site contaminated with ^{60}Co and ^{63}Ni, with ^{60}Co representing 60% of the total activity. The relative fractions are 0.6 for ^{60}Co and 0.4 for ^{63}Ni. If the DCGL for ^{60}Co is 8,300 Bq/m^2 (5,000 dpm/100 cm^2) and the DCGL for ^{63}Ni is 12,000 Bq/m^2 (7,200 dpm/100 cm^2), the gross activity DCGL is 9,500 Bq/m^2 (5,700 dpm/100 cm^2) calculated using Equation 4-4.

When using the gross activity DCGL, it is important to use an appropriately weighted total efficiency to convert from instrument counts to surface activity units using Equations 6-1 through 6-4. In this example, the individual efficiencies for ^{60}Co and ^{63}Ni should be independently evaluated. The overall efficiency is then determined by weighting each individual efficiency by the relative fraction of each radionuclide.

6.6.2 Soil Radionuclide Concentration and Exposure Rates

Analytical procedures, such as alpha and gamma spectrometry, are typically used to determine the radionuclide concentration in soil in units of Bq/kg. Net counts are converted to soil DCGL units by dividing by the time, detector or counter efficiency, mass or volume of the sample, and by the fractional recovery or yield of the chemistry procedure (if applicable). Refer to Chapter 7 for examples of analytical procedures.

Instruments, such as a PIC or micro-R meter, used to measure exposure rate typically read directly in mSv/h. A gamma scintillation detector (*e.g.*, NaI(Tl)) provides data in counts per minute and conversion to mSv/h is accomplished by using site-specific calibration factors developed for the specific instrument (Section 6.5.4).

In situ gamma spectrometry data may require special analysis routines before the spectral data can be converted to soil concentration units or exposure rates.

6.7 Detection Sensitivity

The detection sensitivity of a measurement system refers to a radiation level or quantity of radioactive material that can be measured or detected with some known or estimated level of confidence. This quantity is a factor of both the instrumentation and the technique or procedure being used.

The primary parameters that affect the detection capability of a radiation detector are the background count rate, the detection efficiency of the detector and the counting time interval. It is important to use actual background count rate values and detection efficiencies when determining counting and scanning parameters, particularly during final status and verification surveys. When making field measurements, the detection sensitivity will usually be less than what can be achieved in a laboratory due to increased background and, often times, a significantly lower detection efficiency. It is often impossible to guarantee that pure alpha emitters can be detected *in situ* since the weathering of aged surfaces will often completely absorb the alpha emissions. NRC report NUREG-1507 (NRC 1997b) contains data on many of the parameters that affect detection efficiencies *in situ*, such as absorption, surface smoothness, and particulate radiation energy.

6.7.1 Direct Measurement Sensitivity

Prior to performing field measurements, an investigator must evaluate the detection sensitivity of the equipment proposed for use to ensure that levels below the DCGL can be detected (see Section 4.3). After a direct measurement has been made, it is then necessary to determine whether or not the result can be distinguished from the instrument background response of the measurement system. The terms that are used in this manual to define detection sensitivity for fixed point counts and sample analyses are:

Critical level (L_C)
Detection limit (L_D)
Minimum detectable concentration (MDC)

The critical level (L_C) is the level, in counts, at which there is a statistical probability (with a predetermined confidence) of incorrectly identifying a measurement system background value as "greater than background." Any response above this level is considered to be greater than background. The detection limit (L_D) is an *a priori* estimate of the detection capability of a measurement system, and is also reported in units of counts. The minimum detectable concentration (MDC) is the detection limit (counts) multiplied by an appropriate conversion factor to give units consistent with a site guideline, such as Bq/kg.

The following discussion provides an overview of the derivation contained in the well known publication by Currie (Currie 1968) followed by a description of how the resulting formulae should be used. Publications by Currie (Currie 1968, NRC 1984) and Altshuler and Pasternack (Altshuler and Pasternak 1963) provide details of the derivations involved.

The two parameters of interest for a detector system with a background response greater than zero are:

L_C the net response level, in counts, at which the detector output can be considered "above background"

L_D the net response level, in counts, that can be expected to be seen with a detector with a fixed level of certainty

Assuming that a system has a background response and that random uncertainties and systematic uncertainties are accounted for separately, these parameters can be calculated using Poisson statistics. For these calculations, two types of decision errors should be considered. A Type I error (or "false positive") occurs when a detector response is considered to be above background when, in fact, only background radiation is present. A Type II error (or "false negative") occurs when a detector response is considered to be background when in fact radiation is present at levels above background. The probability of a Type I error is referred to as α (alpha) and is associated with L_C; the probability of a Type II error is referred to as ß (beta) and is associated with L_D. Figure 6.2 graphically illustrates the relationship of these terms with respect to each other and to a normal background distribution.

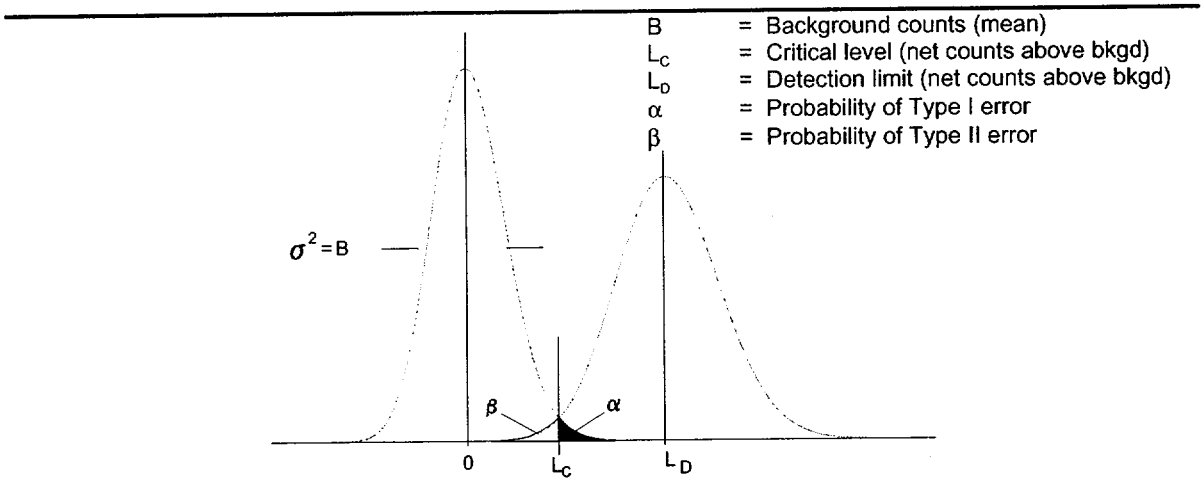

Figure 6.2 Graphically Represented Probabilities for Type I and Type II Errors in Detection Sensitivity for Instrumentation with a Background Response

If α and β are assumed to be equal, the variance (σ^2) of all measurement values is assumed to be equal to the values themselves. If the background of the detection system is not well known, then the critical detection level and the detection limit can be calculated by using the following formulae:

$$L_C = k\sqrt{2B}$$
$$L_D = k^2 + 2k\sqrt{2B} \quad (6\text{-}5)$$

where

L_C = critical level (counts)
L_D = detection limit (counts)
k = Poisson probability sum for α and β (assuming α and β are equal)
B = number of background counts that are expected to occur while performing an actual measurement

The curve to the left in the diagram is the background distribution minus the mean of the background distribution. The result is a Poisson distribution with a mean equal to zero and a variance, σ^2, equal to B. Note that the distribution accounts only for the expected statistical variation due to the stochastic nature of radioactive decay. Currie assumed "paired blanks" when deriving the above stated relationships (Currie 1968), which is interpreted to mean that the sample and background count times are the same.

If values of 0.05 for both α and β are selected as acceptable, then k = 1.645 (from Appendix I, Table I.1) and Equation 6-5 can be written as:

$$L_C = 2.33\sqrt{B}$$
$$L_D = 3 + 4.65\sqrt{B} \quad (6\text{-}6)$$

Note: In Currie's derivation, the constant factor of 3 in the L_D formula was stated as being 2.71, but since that time it has been shown (Brodsky 1992) and generally accepted that a constant factor of 3 is more appropriate. If the sample count times and background count times are different, a slightly different formulation is used.

For an integrated measurement over a preset time, the MDC can be obtained from Equation 6-6 by multiplying by the factor, C. This factor is used to convert from counts to concentration as shown in Equation 6-7:

$$MDC = C \times (3 + 4.65\sqrt{B}) \quad (6\text{-}7)$$

The total detection efficiency and other constants or factors represented by the variable C are usually not truly constants as shown in Equation 6-7. It is likely that at least one of these factors will have a certain amount of variability associated with it which may or may not be significant. These varying factors are gathered together into the single constant, C, by which the net count result will be multiplied when converting the final data. If C varies significantly between measurements, then it might be best to select a value, C′, from the observed distribution of C values that represents a conservative estimate. For example, a value of C might be selected to ensure that at least 95% of the possible values of C are less than the chosen value, C′. The MDC calculated in this way helps assure that the survey results will meet the Data Quality Objectives. This approach for including uncertainties into the MDC calculation is recommended in both NUREG/CR-4007 (NRC 1984) and Appendix A to ANSI N13.30 (ANSI 1996a). Underestimating an MDC can have adverse consequences, especially if activity is later detected at a level above the stated MDC.

Summary of Direct Measurement Sensitivity Terms

- The MDC is the *a priori* net activity level above the critical level that an instrument can be expected to detect 95% of the time. This value should be used when stating the detection capability of an instrument. The MDC is the detection limit, L_D, multiplied by an appropriate conversion factor to give units of activity. Again, this value is used before any measurements are made and is used to estimate the level of activity that can be detected using a given protocol.

- The critical level, L_C, is the lower bound on the 95% detection interval defined for L_D and is the level at which there is a 5% chance of calling a background value "greater than background." This value should be used when actually counting samples or making direct radiation measurements. Any response above this level should be considered as above background (*i.e.*, a net positive result). This will ensure 95% detection capability for L_D.

- From a conservative point of view, it is better to overestimate the MDC for a measurement method. Therefore, when calculating MDC and L_C values, a measurement system background value should be selected that represents the high end of what is expected for a particular measurement method. For direct measurements, probes will be moved from point to point and, as a result, it is expected that the background will most likely vary significantly due to variations in background, source materials, and changes in geometry and shielding. Ideally, the MDC values should be calculated for each type of area, but it may be more economical to simply select a background value from the highest distribution expected and use this for all calculations. For the same reasons, realistic values of detection efficiencies and other process parameters should be used when possible and should be reflective of the actual conditions. To a great degree, the selection of these parameters will be based on judgment and will require evaluation of site-specific conditions.

MDC values for other counting conditions may be derived from Equation 6-7 depending on the detector and contaminants of concern. For example, it may be required to determine what level of contamination, distributed over 100 cm^2, can be detected with a 500 cm^2 probe or what contamination level can be detected with any probe when the contamination area is smaller than the probe active area. Table 6.4 lists several common field survey detectors with estimates of MDC values for ^{238}U on a smooth, flat plane. As such, these represent minimum MDC values and may not be applicable at all sites. Appropriate site-specific MDC values should be determined using the DQO Process.

Table 6.4 Examples of Estimated Detection Sensitivities for Alpha and Beta Survey Instrumentation

(Static one minute counts for ^{238}U calculated using Equations 6-6 and 6-7)

Detector	Probe area (cm^2)	Background (cpm)	Efficiency (cpm/dpm)	Approximate Sensitivity		
				L_C (counts)	L_D (counts)	MDC (Bq/m^2)[a]
Alpha proportional	50	1	0.15	2	7	150
Alpha proportional	100	1	0.15	2	7	83
Alpha proportional	600	5	0.15	5	13	25
Alpha scintillation	50	1	0.15	2	7	150
Beta proportional	100	300	0.20	40	83	700
Beta proportional	600	1500	0.20	90	183	250
Beta GM pancake	15	40	0.20	15	32	1800

[a] Assumes that the size of the contamination area is at least as large as the probe area.

Sample Calculation 1:

The following example illustrates the calculation of an MDC in Bq/m^2 for an instrument with a 15 cm^2 probe area when the measurement and background counting times are each one minute:

B = 40 counts
C = (5 dpm/count)(Bq/60 dpm)(1/15 cm^2 probe area)(10,000 cm^2/m^2)
= 55.6 Bq/m^2-counts

The MDC is calculated using Equation 6-7:

$$MDC = 55.6 \times (3 + 4.65\sqrt{40}) = 5{,}000\ Bq/m^2\ (3{,}000\ dpm/100\ cm^2)$$

The critical level, L_c, for this example is calculated from Equation 6-6:

$$L_C = 2.33\sqrt{B} = 15 \text{ counts}$$

Given the above scenario, if a person asked what level of contamination could be detected 95% of the time using this method, the answer would be 5,000 Bq/m^2 (3,000 dpm/100 cm^2). When actually performing measurements using this method, any count yielding greater than 55 total counts, or greater than 15 net counts (55-40=15) during a period of one minute, would be regarded as greater than background.

6.7.2 Scanning Sensitivity

The ability to identify a small area of elevated radioactivity during surface scanning is dependent upon the surveyor's skill in recognizing an increase in the audible or display output of an instrument. For notation purposes, the term "scanning sensitivity" is used throughout this section to describe the ability of a surveyor to detect a pre-determined level of contamination with a detector. The greater the sensitivity, the lower the level of contamination that can be detected.

Many of the radiological instruments and monitoring techniques typically used for occupational health physics activities may not provide the detection sensitivities necessary to demonstrate compliance with the DCGLs. The detection sensitivity for a given application can be improved (*i.e.*, lower the MDC) by: 1) selecting an instrument with a higher detection efficiency or a lower background, 2) decreasing the scanning speed, or 3) increasing the size of the effective probe area without significantly increasing the background response.

Scanning is usually performed during radiological surveys in support of decommissioning to identify the presence of any areas of elevated activity. The probability of detecting residual contamination in the field depends not only on the sensitivity of the survey instrumentation when used in the scanning mode of operation, but is also affected by the surveyor's ability—*i.e.*, human factors. The surveyor must make a decision whether the signals represent only the background

activity, or residual contamination in excess of background. The greater the sensitivity, the lower the level of contamination that may be detected by scanning. Accounting for these human factors represents a significant change from the traditionally accepted methods of estimating scanning sensitivities.

An empirical method for evaluating the detection sensitivity for contamination surveys is by actual experimentation or, since it is certainly feasible, by simulating an experimental setup using computer software. The following steps provide a simple example of how one can perform this empirical evaluation:

1) A desired nuclide contamination level is selected.
2) The response of the detector to be used is determined for the selected nuclide contamination level.
3) A test source is constructed which will give a detector count rate equivalent to what was determined in step 2. The count rate is equivalent to what would be expected from the detector when placed on an actual contamination area equal in value to that selected in step 1.
4) The detector of choice is then moved over the source at different scan rates until an acceptable speed is determined.

The most useful aspect of this approach is that the source can then be used to show surveyors what level of contamination is expected to be targeted with the scan. They, in turn, can gain experience with what the expected response of the detector will be and how fast they can survey and still feel comfortable about detecting the target contamination level. The person responsible for the survey can then use this information when developing a fixed point measurement and sampling plan.

The remainder of this section is dedicated to providing the reader with information pertaining to the underlying processes involved when performing scanning surveys for alpha, beta, and gamma emitting radionuclides. The purpose is to provide relevant information that can be used for estimating realistic scanning sensitivities for survey activities.

6.7.2.1 Scanning for Beta and Gamma Emitters

The minimum detectable concentration of a scan survey (scan MDC) depends on the intrinsic characteristics of the detector (efficiency, physical probe area, *etc.*), the nature (type and energy of emissions) and relative distribution of the potential contamination (point versus distributed source and depth of contamination), scan rate, and other characteristics of the surveyor. Some factors that may affect the surveyor's performance include the costs associated with various outcomes—*e.g.*, fatigue, noise, level of training, experience—and the survey's *a priori* expectation of the likelihood of contamination present. For example, if the surveyor believes that

the potential for contamination is very low, as in a Class 3 area, a relatively large signal may be required for the surveyor to conclude that contamination is present. NRC draft report NUREG/CR-6364 (NRC 1997d) provides a complete discussion of the human factors as they relate to the performance of scan surveys.

Signal Detection Theory. Personnel conducting radiological surveys for residual contamination at decommissioning sites must interpret the audible output of a portable survey instrument to determine when the signal ("clicks") exceeds the background level by a margin sufficient to conclude that contamination is present. It is difficult to detect low levels of contamination because both the signal and the background vary widely. Signal detection theory provides a framework for the task of deciding whether the audible output of the survey meter during scanning is due to background or signal plus background levels. An index of sensitivity (d') that represents the distance between the means of the background and background plus signal (refer to Figure 6.2 for determining L_D), in units of their common standard deviation, can be calculated for various decision errors (correct detection and false positive rate). As an example, for a correct detection rate of 95% (complement of a false negative rate of 5%) and a false positive rate of 5%, d' is 3.29 (similar to the static MDC for the same decision error rates). The index of sensitivity is independent of human factors, and therefore, the ability of an ideal observer (theoretical construct), may be used to determine the minimum d' that can be achieved for particular decision errors. The ideal observer makes optimal use of the available information to maximize the percent correct responses, providing an effective upper bound against which to compare actual surveyors. Table 6.5 lists selected values of d'.

Two Stages of Scanning. The framework for determining the scan MDC is based on the premise that there are two stages of scanning. That is, surveyors do not make decisions on the basis of a single indication, rather, upon noting an increased number of counts, they pause briefly and then decide whether to move on or take further measurements. Thus, scanning consists of two components: continuous monitoring and stationary sampling. In the first component, characterized by continuous movement of the probe, the surveyor has only a brief "look" at potential sources, determined by the scan speed. The surveyor's willingness to decide that a signal is present at this stage is likely to be liberal, in that the surveyor should respond positively on scant evidence, since the only "cost" of a false positive is a little time. The second component occurs only after a positive response was made at the first stage. This response is marked by the surveyor interrupting his scanning and holding the probe stationary for a period of time, while comparing the instrument output signal during that time to the background counting rate. Owing to the longer observation interval, sensitivity is relatively high. For this decision, the criterion should be more strict, since the cost of a "yes" decision is to spend considerably more time taking a static measurement or a sample.

Table 6.5 Values of d' for Selected True Positive and False Positive Proportions

False Positive Proportion	True Positive Proportion							
	0.60	**0.65**	**0.70**	**0.75**	**0.80**	**0.85**	**0.90**	**0.95**
0.05	1.90	2.02	2.16	2.32	2.48	2.68	2.92	3.28
0.10	1.54	1.66	1.80	1.96	2.12	2.32	2.56	2.92
0.15	1.30	1.42	1.56	1.72	1.88	2.08	2.32	2.68
0.20	1.10	1.22	1.36	1.52	1.68	1.88	2.12	2.48
0.25	0.93	1.06	1.20	1.35	1.52	1.72	1.96	2.32
0.30	0.78	0.91	1.05	1.20	1.36	1.56	1.80	2.16
0.35	0.64	0.77	0.91	1.06	1.22	1.42	1.66	2.02
0.40	0.51	0.64	0.78	0.93	1.10	1.30	1.54	1.90
0.45	0.38	0.52	0.66	0.80	0.97	1.17	1.41	1.77
0.50	0.26	0.38	0.52	0.68	0.84	1.04	1.28	1.64
0.55	0.12	0.26	0.40	0.54	0.71	0.91	1.15	1.51
0.60	0.00	0.13	0.27	0.42	0.58	0.82	1.02	1.38

Since scanning can be divided into two stages, it is necessary to consider the survey's scan sensitivity for each of the stages. Typically, the minimum detectable count rate (MDCR) associated with the first scanning stage will be greater due to the brief observation intervals of continuous monitoring—provided that the length of the pause during the second stage is significantly longer. Typically, observation intervals during the first stage are on the order of 1 or 2 seconds, while the second stage pause may be several seconds long. The greater value of MDCR from each of the scan stages is used to determine the scan sensitivity for the surveyor.

Determination of MDCR and Use of Surveyor Efficiency. The minimum detectable number of net source counts in the interval is given by s_i. Therefore, for an ideal observer, the number of source counts required for a specified level of performance can be arrived at by multiplying the square root of the number of background counts by the detectability value associated with the desired performance (as reflected in d') as shown in Equation 6-8:

$$s_i = d' \sqrt{b_i} \qquad (6\text{-}8)$$

where the value of d' is selected from Table 6.5 based on the required true positive and false positive rates and b_i is the number of background counts in the interval.

For example, suppose that one wished to estimate the minimum count rate that is detectable by scanning in an area with a background of 1,500 cpm. Note that the minimum detectable count rate must be considered for both scan stages—and the more conservative value is selected as the minimum count rate that is detectable. It will be assumed that a typical source remains under the probe for 1 second during the first stage, therefore, the average number of background counts in the observation interval is 25 ($b_i = 1500 \times (1/60)$). Furthermore, as explained earlier, it can be assumed that at the first scanning stage a high rate (*e.g.*, 95%) of correct detections is required, and that a correspondingly high rate of false positives (*e.g.*, 60%) will be tolerated. From Table 6.5, the value of d', representing this performance goal, is 1.38. The net source counts needed to support the specified level of performance (assuming an ideal observer) will be estimated by multiplying 5 (the square root of 25) by 1.38. Thus, the net source counts per interval, s_i, needed to yield better than 95% detections with about 60% false positives is 6.9. The minimum detectable source count rate, in cpm, may be calculated by:

$$MDCR = s_i \times (60/i) \tag{6-9}$$

For this example, MDCR is equivalent to 414 cpm (1,914 cpm gross). Table 6.6 provides the scan sensitivity for the ideal observer (MDCR) at the first scanning stage for various background levels, based on an index of sensitivity (d') of 1.38 and a 2-second observation interval.

Table 6.6 Scanning Sensitivity (MDCR) of the Ideal Observer for Various Background Levels[a]

Background (cpm)	MDCR (net cpm)	Scan Sensitivity (gross cpm)
45	50	95
60	60	120
260	120	380
300	130	430
350	140	490
400	150	550
1,000	240	1,240
3,000	410	3,410
4,000	480	4,480

*The sensitivity of the ideal observer during the first scanning stage is based on an index of sensitivity (d') of 1.38 and a 2-second observation interval.

The minimum number of source counts required to support a given level of performance for the final detection decision (second scan stage) can be estimated using the same method. As explained earlier, the performance goal at this stage will be more demanding. The required rate of true positives remains high (*e.g.*, 95%), but fewer false positives (*e.g.*, 20%) can be tolerated, such that d' (from Table 6.5) is now 2.48. One will assume that the surveyor typically stops the probe over a suspect location for about 4 seconds before making a decision, so that the average number of background counts in an observation interval is 100 ($b_i = 1{,}500 \times (4/60)$). Therefore, the minimum detectable number of net source counts, s_i, needed will be estimated by multiplying 10 (the square root of 100) by 2.48 (the d' value); so s_i equals 24.8. The MDCR is calculated by $2.48 \times (60/4)$ and equals 372 cpm. The value associated with the first scanning stage (this example, 414 cpm) will typically be greater, owing to the relatively brief intervals assumed.

Laboratory studies using simulated sources and backgrounds were performed to assess the abilities of surveyors under controlled conditions. The methodology and analysis of results for these studies are described in draft NUREG/CR-6364 (NRC 1997d) and NUREG-1507 (NRC 1997b). The surveyor's actual performance as compared with that which is ideally possible (using the ideal observer construct) provided an indication of the efficiency of the surveyors. Based on the results of the confidence rating experiment, this surveyor efficiency (p) was estimated to be between 0.5 and 0.75.

MARSSIM recommends assuming an efficiency value at the lower end of the observed range (*i.e.*, 0.5) when making MDC estimates. Thus, the required number of net source counts for the surveyor, $MDCR_{surveyor}$, is determined by dividing the MDCR by the square root of p. Continuing with this example, the surveyor MDCR is calculated by 414 cpm/0.707, or 585 cpm (2,085 cpm gross).

Scan MDCs for Structure Surfaces and Land Areas. The survey design for determining the number of data points for areas of elevated activity (see Section 5.5.2.4) depends on the scan MDC for the selected instrumentation. In general, alpha or beta scans are performed on structure surfaces to satisfy the elevated activity measurements survey design, while gamma scans are performed for land areas. Because of low background levels for alpha emitters, the approach described here is not generally applied to determining scan MDCs for alpha contaminants—rather, the reader is referred to Section 6.7.2.2 for an appropriate method for determining alpha scan MDCs for building surfaces. In any case, the data requirements for assessing potential elevated areas of direct radiation depend on the scan MDC of the survey instrument (*e.g.*, floor monitor, GM detector, NaI scintillation detector).

Scan MDCs for Building/Structure Surfaces. The scan MDC is determined from the minimum detectable count rate (MDCR) by applying conversion factors that account for detector and surface characteristics and surveyor efficiency. As discussed above, the MDCR accounts for the background level, performance criteria (d'), and observation interval. The observation interval

during scanning is the actual time that the detector can respond to the contamination source—this interval depends on the scan speed, detector size in the direction of the scan, and area of elevated activity. Because the actual dimensions of potential areas of elevated activity in the field cannot be known *a priori*, MARSSIM recommends postulating a certain area (*e.g.*, perhaps 50 to 200 cm^2), and then selecting a scan rate that provides a reasonable observation interval.

Finally, the scan MDC for structure surfaces may be calculated:

$$Scan\,MDC = \frac{MDCR}{\sqrt{p}\,\varepsilon_i\,\varepsilon_s\,\frac{probe\,area}{100\,cm^2}} \qquad (6\text{-}10)$$

where

MDCR	=	minimum detectable count rate
ε_i	=	instrument efficiency
ε_s	=	surface efficiency
p	=	surveyor efficiency

As an example, the scan MDC (in dpm/100 cm^2) for ^{99}Tc on a concrete surface may be determined for a background level of 300 cpm and a 2-second observation interval using a hand-held gas proportional detector (126 cm^2 probe area). For a specified level of performance at the first scanning stage of 95% true positive rate and 60% false positive rate (and assuming the second stage pause is sufficiently long to ensure that the first stage is more limiting), d' equals 1.38 (Table 6.5) and the MDCR is 130 cpm (Table 6.6). Using a surveyor efficiency of 0.5, and assuming instrument and surface efficiencies of 0.36 and 0.54, respectively, the scan MDC is calculated using Equation 6-10:

$$Scan\ MDC = \frac{130}{\sqrt{0.5}\ (0.36)\ (0.54)\ (1.26)} = 750\ dpm/100\ cm^2$$

Additional examples for calculating the scan MDC may be found in NUREG-1507 (NRC 1997b).

Scan MDCs for Land Areas. In addition to the MDCR and detector characteristics, the scan MDC (in pCi/g) for land areas is based on the area of elevated activity, depth of contamination, and the radionuclide (*i.e.*, energy and yield of gamma emissions). If one assumes constant parameters for each of the above variables, with the exception of the specific radionuclide in question, the scan MDC may be reduced to a function of the radionuclide alone. NaI scintillation detectors are generally used for scanning land areas.

An overview of the approach used to determine scan MDCs for land areas follows. The NaI(Tl) scintillation detector background level and scan rate (observation interval) are postulated, and the MDCR for the ideal observer, for a given level of performance, is obtained. After a surveyor efficiency is selected, the relationship between the surveyor MDCR ($MDCR_{surveyor}$) and the radionuclide concentration in soil (in Bq/kg or pCi/g)is determined. This correlation requires two steps—first, the relationship between the detector's net count rate to net exposure rate (cpm per μR/h) is established, and second, the relationship between the radionuclide contamination and exposure rate is determined.

For a particular gamma energy, the relationship of NaI(Tl) scintillation detector count rate and exposure rate may be determined analytically (in cpm per μR/h). The approach used to determine the gamma fluence rate necessary to yield a fixed exposure rate (1 μR/h)—as a function of gamma energy—is provided in NUREG-1507 (NRC 1997b). The NaI(Tl) scintillation detector response (cpm) is related to the fluence rate at specific energies, considering the detector's efficiency (probability of interaction) at each energy. From this, the NaI(Tl) scintillation detector versus exposure rates for varying gamma energies are determined. Once the relationship between the NaI(Tl) scintillation detector response (cpm) and the exposure rate is established, the $MDCR_{surveyor}$ (in cpm) of the NaI(Tl) scintillation detector can be related to the minimum detectable net exposure rate. The minimum detectable exposure rate is used to determine the minimum detectable radionuclide concentration (*i.e.*, the scan MDC) by modeling a specified small area of elevated activity.

Modeling (using Microshield™) of the small area of elevated activity (soil concentration) is used to determine the net exposure rate produced by a radionuclide concentration at a distance 10 cm above the source. This position is selected because it relates to the average height of the NaI(Tl) scintillation detector above the ground during scanning.

The factors considered in the modeling include:

- radionuclide of interest (considering all gamma emitters for decay chains)
- expected concentration of the radionuclide of interest
- areal dimensions of the area of elevated activity
- depth of the area of elevated activity
- location of dose point (NaI(Tl) scintillation detector height above the surface)
- density of soil

Modeling analyses are conducted by selecting a radionuclide (or radioactive material decay series) and then varying the concentration of the contamination. The other factors are held constant—the areal dimension of a cylindrical area of elevated activity is 0.25 m^2 (radius of 28 cm), the depth of the area of elevated activity is 15 cm, the dose point is 10 cm above the surface, and the density of soil is 1.6 g/cm^3. The objective is to determine the radionuclide concentration that is correlated to the minimum detectable net exposure rate.

As an example, the scan MDC for ^{137}Cs using a 1.5 in. by 1.25 in. NaI(Tl) scintillation detector is considered in detail. Assume that the background level is 4,000 cpm and that the desired level of performance, 95% correct detections and 60% false positive rate, results in a d' of 1.38. The scan rate of 0.5m/s provides an observation interval of 1-second (based on a diameter of about 56 cm for the area of elevated activity). The $MDCR_{surveyor}$ may be calculated assuming a surveyor efficiency (p) of 0.5 as follows:

1) $b_i = (4{,}000 \text{ cpm}) \times (1 \text{ sec}) \times (1 \text{ min}/60 \text{ sec}) = 66.7 \text{ counts}$

2) $MDCR = (1.38) \times (\sqrt{66.7}) \times (60 \text{ sec}/1 \text{ min}) = 680 \text{ cpm}$

3) $MDCR_{surveyor} = 680/\sqrt{0.5} = 960 \text{ cpm}$

The corresponding minimum detectable exposure rate is determined for this detector and radionuclide. The manufacturer of this particular 1.5 in. by 1.25 in. NaI(Tl) scintillation detector quotes a count rate to exposure rate ratio for ^{137}Cs of 350 cpm per μR/h. The minimum detectable exposure rate is calculated by dividing the count rate (960 cpm) by the count rate to exposure rate ratio for the radionuclide of interest (350 cpm per μR/h). The minimum detectable exposure rate for this example is 2.73 μR/h.

Both ^{137}Cs and its short-lived progeny, ^{137m}Ba, were chosen from the Microshield™ library. The source activity and other modeling parameters were entered into the modeling code. The source activity was selected based on an arbitrary concentration of 5 pCi/g. The modeling code performed the appropriate calculations and determined an exposure rate of 1.307 μR/h (which accounts for buildup). Finally, the radionuclide concentrations of ^{137}Cs and ^{137m}Ba (scan MDC) necessary to yield the minimum detectable exposure rate (2.73 μR/h) may be calculated using the following formula.

$$scan\,MDC = \frac{(5\ pCi/g)(2.73\ \mu R/h)}{1.307\ \mu R/h} = 10.4\ pCi/g \qquad (6\text{-}11)$$

It must be emphasized that while a single scan MDC value can be calculated for a given radionuclide—other scan MDC values may be equally justifiable depending on the values chosen for the various factors, including the MDCR (background level, acceptable performance criteria, observation interval), surveyor efficiency, detector parameters and the modeling conditions of the contamination. It should also be noted that determination of the scan MDC for radioactive materials—like uranium and thorium—must consider the gamma radiation emitted from the entire decay series. NUREG-1507 (NRC 1997b) provides a detailed example of how the scan MDC can be determined for enriched uranium.

Table 6.7 provides scan MDCs for common radionuclides and radioactive materials in soil. It is important to note that the variables used in the above examples to determine the scan MDCs for the 1.25 in. by 1.5 in. NaI(Tl) scintillation detector—*i.e.*, the $MDCR_{surveyor}$ detector parameters (*e.g.*, cpm per μR/h), and the characteristics of the area of elevated activity—have all been held constant to facilitate the calculation of scan MDCs provided in Table 6.7. The benefit of this approach is that generally applicable scan MDCs are provided for different radioactive contaminants. Additionally, the relative detectability of different contaminants is evident because the only variable in Table 6.7 is the nature of the contaminant.

As noted above, the scan MDCs calculated using the approach in this section are dependent on several factors. One way to validate the appropriateness of the scan MDC is by tracking the residual radioactivity (both surface activity and soil concentrations) levels identified during investigations performed as a result of scanning surveys. The measurements performed during these investigations may provide an *a posteriori* estimate of the scan MDC that can be used to validate the *a priori* scan MDC used to design the survey.

6.7.2.2 Scanning for Alpha Emitters

Scanning for alpha emitters differs significantly from scanning for beta and gamma emitters in that the expected background response of most alpha detectors is very close to zero. The following discussion covers scanning for alpha emitters and assumes that the surface being surveyed is similar in nature to the material on which the detector was calibrated. In this respect, the approach is purely theoretical. Surveying surfaces that are dirty, non-planar, or weathered can significantly affect the detection efficiency and therefore bias the expected MDC for the scan. The use of reasonable detection efficiency values instead of optimistic values is highly recommended. Appendix J contains a complete derivation of the alpha scanning equations used in this section.

Since the time a contaminated area is under the probe varies and the background count rate of some alpha instruments is less than 1 cpm, it is not practical to determine a fixed MDC for scanning. Instead, it is more useful to determine the probability of detecting an area of contamination at a predetermined DCGL for given scan rates.

For alpha survey instrumentation with backgrounds ranging from <1 to 3 cpm, a single count provides a surveyor sufficient cause to stop and investigate further. Assuming this to be true, the probability of detecting given levels of alpha surface contamination can be calculated by use of Poisson summation statistics.

Table 6.7 NaI(Tl) Scintillation Detector Scan MDCs for Common Radiological Contaminants[a]

Radionuclide/Radioactive Material	1.25 in. by 1.5 in. NaI Detector		2 in. by 2 in. NaI Detector	
	Scan MDC (Bq/kg)	Weighted cpm/μR/h	Scan MDC (Bq/kg)	Weighted cpm/μR/h
Am-241	1,650	5,830	1,170	13,000
Co-60	215	160	126	430
Cs-137	385	350	237	900
Th-230	111,000	4,300	78,400	9,580
Ra-226 (in equilibrium with progeny)	167	300	104	760
Th-232 decay series (Sum of all radionuclides in he thorium decay series)	1,050	340	677	830
Th-232 (In equilibrium with progeny in decay series)	104	340	66.6	830
Depleted Uranium[b] (0.34% U-235)	2,980	1,680	2,070	3,790
Natural Uranium[b]	4,260	1,770	2,960	3,990
3% Enriched Uranium[b]	5,070	2,010	3,540	4,520
20% Enriched Uranium[b]	5,620	2,210	3,960	4,940
50% Enriched Uranium[b]	6,220	2,240	4,370	5,010
75% Enriched Uranium[b]	6,960	2,250	4,880	5,030

[a] Refer to text for complete explanation of factors used to calculate scan MDCs. For example, the background level for the 1.25 in. by 1.5 in. NaI detector was assumed to be 4,000 cpm, and 10,000 cpm for the 2 in. by 2 in. NaI detector. The observation interval was 1-sec and the level of performance was selected to yield d' of 1.38.

[b] Scan MDC for uranium includes sum of ^{238}U, ^{235}U, and ^{234}U.

Given a known scan rate and a surface contamination DCGL, the probability of detecting a single count while passing over the contaminated area is

$$P(n \geq 1) = 1 - e^{\frac{-GEd}{60v}} \qquad (6\text{-}12)$$

where

$P(n \geq 1)$	=	probability of observing a single count
G	=	contamination activity (dpm)
E	=	detector efficiency (4π)
d	=	width of detector in direction of scan (cm)
v	=	scan speed (cm/s)

Note: Refer to Appendix J for a complete derivation of these formulas.

Once a count is recorded and the guideline level of contamination is present the surveyor should stop and wait until the probability of getting another count is at least 90%. This time interval can be calculated by

$$t = \frac{13{,}800}{CAE} \qquad (6\text{-}13)$$

where

t	=	time period for static count (s)
C	=	contamination guideline (dpm/100 cm^2)
A	=	physical probe area (cm^2)
E	=	detector efficiency (4π)

Many portable proportional counters have background count rates on the order of 5 to 10 cpm, and a single count should not cause a surveyor to investigate further. A counting period long enough to establish that a single count indicates an elevated contamination level would be prohibitively inefficient. For these types of instruments, the surveyor usually will need to get at least 2 counts while passing over the source area before stopping for further investigation.

Assuming this to be a valid assumption, the probability of getting two or more counts can be calculated by:

$$P(n\geq 2) = 1 - P(n=0) - P(n=1)$$

$$= 1 - \left(1 + \frac{(GE + B)t}{60}\right)\left(e^{-\frac{(GE + B)t}{60}}\right) \qquad (6\text{-}14)$$

where

$P(n\geq 2)$	=	probability of getting 2 or more counts during the time interval t
$P(n=0)$	=	probability of not getting any counts during the time interval t
$P(n=1)$	=	probability of getting 1 count during the time interval t
B	=	background count rate (cpm)

All other variables are the same as for Equation 6-12.

Appendix J provides a complete derivation of Equations 6-12 through 6-14 and a detailed discussion of the probability of detecting alpha surface contamination for several different variables. Several probability charts are included at the end of Appendix J for common detector sizes. Table 6.8 provides estimates of the probability of detecting 300 dpm/100 cm^2 for some commonly used alpha detectors.

Table 6.8 Probability of Detecting 300 dpm/100 cm^2 of Alpha Activity While Scanning with Alpha Detectors Using an Audible Output (calculated using Equation 6-12)

Detector Type	Detection Efficiency cpm/dpm	Probe Dimension in Direction of Scan (cm)	Scan Rate (cm/s)	Probability of detecting 300 dpm/100 cm^2
Proportional	0.20	5	3	80%
Proportional	0.15	15	5	90%
Scintillation	0.15	5	3	70%
Scintillation	0.15	10	3	90%

6.8 Measurement Uncertainty (Error)

The quality of measurement data will be directly impacted by the magnitude of the measurement uncertainty associated with it. Some uncertainties, such as statistical counting uncertainties, can be easily calculated from the count results using mathematical procedures. Evaluation of other

sources of uncertainty require more effort and in some cases is not possible. For example, if an alpha measurement is made on a porous concrete surface, the observed instrument response when converted to units of activity will probably not exactly equal the true activity under the probe. Variations in the absorption properties of the surface for particulate radiation will vary from point to point and therefore will create some level of variation in the expected detection efficiency. This variability in the expected detector efficiency results in uncertainty in the final reported result. In addition, QC measurement results provide an estimate of random and systematic uncertainties associated with the measurement process.

The measurement uncertainty for every analytical result or series of results, such as for a measurement system, should be reported. This uncertainty, while not directly used for demonstrating compliance with the release criterion, is used for survey planning and data assessment throughout the Radiation Survey and Site Investigation (RSSI) process. In addition, the uncertainty is used for evaluating the performance of measurement systems using QC measurement results. Uncertainty can also be used for comparing individual measurements to the DCGL. This is especially important in the early stages of decommissioning (*i.e.*, scoping, characterization, remedial action support) when decisions are made based on a limited number of measurements.

For most sites, evaluations of uncertainty associated with field measurements is important only for data being used as part of the final status survey documentation. The final status survey data, which is used to document the final radiological status of a site, should state the uncertainties associated with the measurements. Conversely, detailing the uncertainties associated with measurements made during scoping or characterization surveys may or may not be of value depending on what the data will be used for—*i.e.* the data quality objectives (DQOs). From a practical standpoint, if the observed data are obviously greater than the DCGL and will be eventually cleaned up, then the uncertainty may be relatively unimportant. Conversely, data collected during early phases of a site investigation that may eventually be used to show that the area is below the DCGL—and therefore does not require any clean-up action—will need the same uncertainty evaluation as the final status survey data. In summary, the level of effort needs to match the intended use of the data.

6.8.1 Systematic and Random Uncertainties

Measurement uncertainties are often broken into two sub-classes of uncertainty termed systematic (*e.g.*, methodical) uncertainty and random (*e.g.*, stochastic) uncertainty. Systematic uncertainties derive from a lack of knowledge about the true distribution of values associated with a numerical parameter and result in data that is consistently higher (or lower) than the true value. An example of a systematic uncertainty would be the use of a fixed counting efficiency value even though it is known that the efficiency varies from measurement to measurement but without knowledge of the frequency. If the fixed counting efficiency value is higher than the true but unknown

efficiency—as would be the case for an unrealistically optimistic value—then every measurement result calculated using that efficiency would be biased low. Random uncertainties refer to fluctuations associated with a known distribution of values. An example of a random uncertainty would be a well documented chemical separation efficiency that is known to fluctuate with a regular pattern about a mean. A constant recovery value is used during calculations, but the true value is known to fluctuate from sample to sample with a fixed and known degree of variation.

To minimize the need for estimating potential sources of uncertainty, the sources of uncertainty themselves should be reduced to a minimal level by using practices such as:

- The detector used should minimize the potential uncertainty. For example, when making field surface activity measurements for ^{238}U on concrete, a beta detector such as a thin-window Geiger-Mueller "pancake" may provide better quality data than an alpha detector depending on the circumstances. Less random uncertainty would be expected between measurements with a beta detector such as a pancake since beta emissions from the uranium will be affected much less by thin absorbent layers than will the alpha emissions.

- Calibration factors should accurately reflect the efficiency of a detector being used on the surface material being measured for the contaminant radionuclide or mixture of radionuclides (see Section 6.5.4). For most field measurements, variations in the counting efficiency on different types of materials will introduce the largest amount of uncertainty in the final result.

- Uncertainties should be reduced or eliminated by use of standardized measurement protocols (*e.g.*, SOPs) when possible. Special effort should be made to reduce or eliminate systematic uncertainties, or uncertainties that are the same for every measurement simply due to an error in the process. If the systematic uncertainties are reduced to a negligible level, then the random uncertainties, or those uncertainties that occur on a somewhat statistical basis, can be dealt with more easily.

- Instrument operators should be trained and experienced with the instruments used to perform the measurements.

- QA/QC should be conducted as described in Chapter 9.

Uncertainties that cannot be eliminated need to be evaluated such that the effect can be understood and properly propagated into the final data and uncertainty estimates. As previously stated, non-statistical uncertainties should be minimized as much as possible through the use of good work practices.

Overall random uncertainty can be evaluated using the methods described in the following sections. Section 6.8.2 describes a method for calculating random counting uncertainty. Section 6.8.3 discusses how to combine this counting uncertainty with other uncertainties from the measurement process using uncertainty propagation.

Systematic uncertainty is derived from calibration errors, incorrect yields and efficiencies, non-representative survey designs, and "blunders." It is difficult—and sometimes impossible—to evaluate the systematic uncertainty for a measurement process, but bounds should always be estimated and made small compared to the random uncertainty, if possible. If no other information on systematic uncertainty is available, Currie (NRC 1984) recommends using 16% as an estimate for systematic uncertainties (1% for blanks, 5% for baseline, and 10% for calibration factors).

6.8.2 Statistical Counting Uncertainty

When performing an analysis with a radiation detector, the result will have an uncertainty associated with it due to the statistical nature of radioactive decay. To calculate the total uncertainty associated with the counting process, both the background measurement uncertainty and the sample measurement uncertainty must be accounted for. The standard deviation of the net count rate, or the statistical counting uncertainty, can be calculated by

$$\sigma_n = \sqrt{\frac{C_{s+b}}{T_{s+b}^2} + \frac{C_b}{T_b^2}} \qquad (6\text{-}15)$$

where

σ_n = standard deviation of the net count rate result
C_{s+b} = number of gross counts (sample)
T_{s+b} = gross count time
C_b = number of background counts
T_b = background count time

6.8.3 Uncertainty Propagation

Most measurement data will be converted to different units or otherwise included in a calculation to determine a final result. The standard deviation associated with the final result, or the total uncertainty, can then be calculated. Assuming that the individual uncertainties are relatively small, symmetric about zero, and independent of one another, then the total uncertainty for the final calculated result can be determined by solving the following partial differential equation:

$$\sigma_u = \sqrt{\left(\frac{\delta u}{\delta x}\right)^2 \sigma_x^2 + \left(\frac{\delta u}{\delta y}\right)^2 \sigma_y^2 + \left(\frac{\delta u}{\delta z}\right)^2 \sigma_z^2 + \ldots} \qquad (6\text{-}16)$$

where

u	=	function, or formula, that defines the calculation of a final result as a function of the collected data. All variables in this equation, *i.e.*, x, y, z..., are assumed to have a measurement uncertainty associated with them and do not include numerical constants
σ_u	=	standard deviation, or uncertainty, associated with the final result
$\sigma_x, \sigma_y, \ldots$	=	standard deviation, or uncertainty, associated with the parameters x, y, z, ...

Equation 6-16, generally known as the error propagation formula, can be solved to determine the standard deviation of a final result from calculations involving measurement data and their associated uncertainties. The solutions for common calculations along with their uncertainty propagation formulas are included below.

Data Calculation	Uncertainty Propagation
$u = x + y$, or $u = x - y$:	$\sigma_u = \sqrt{\sigma_x^2 + \sigma_y^2}$
$u = x \div y$, or $u = x \times y$:	$\sigma_u = u\sqrt{\left(\frac{\sigma_x}{x}\right)^2 + \left(\frac{\sigma_y}{y}\right)^2}$
$u = c \times x$, where c is a positive constant:	$\sigma_u = c\,\sigma_x$
$u = x \div c$, where c is a positive constant:	$\sigma_u = \frac{\sigma_x}{c}$

Note: In the above examples, x and y are measurement values with associated standard deviations, or uncertainties, equal to σ_x and σ_y respectively. The symbol "c" is used to represent a numerical constant which has no associated uncertainty. The symbol σ_u is used to denote the standard deviation, or uncertainty, of the final calculated value u.

6.8.4 Reporting Confidence Intervals

Throughout Section 6.8, the term "measurement uncertainty" is used interchangeably with the term "standard deviation." In this respect, the uncertainty is qualified as numerically identical to

the standard deviation associated with a normally distributed range of values. When reporting a confidence interval for a value, one provides the range of values that represent a pre-determined level of confidence (*i.e.*, 95%). To make this calculation, the final standard deviation, or total uncertainty σ_u as shown in Equation 6-16, is multiplied by a constant factor k representing the area under a normal curve as a function of the standard deviation. The values of k representing various intervals about a mean of normal distributions as a function of the standard deviation is given in Table 6.9. The following example illustrates the use of this factor in context with the propagation and reporting of uncertainty values.

Table 6.9 Areas Under Various Intervals About the Mean of a Normal Distribution

Interval ($\bar{\mu} \pm k\sigma$)	Area
$\bar{\mu} \pm 0.674\sigma$	0.500
$\bar{\mu} \pm 1.00\sigma$	0.683
$\bar{\mu} \pm 1.65\sigma$	0.900
$\bar{\mu} \pm 1.96\sigma$	0.950
$\bar{\mu} \pm 2.00\sigma$	0.954
$\bar{\mu} \pm 2.58\sigma$	0.990
$\bar{\mu} \pm 3.00\sigma$	0.997

Example:

Uncertainty Propagation and Confidence Interval: A measurement process with a zero background yields a count result of 28 ± 5 counts in 5 minutes, where the ± 5 counts represents one standard deviation about a mean value of 28 counts. The detection efficiency is 0.1 counts per disintegration ± 0.01 counts per disintegration, again representing one standard deviation about the mean.

Calculate the activity of the sample, in dpm, total measurement uncertainty, and the 95% confidence interval for the result.

1) The total number of disintegrations is:

$$\frac{28\ counts}{0.1\ c/d} = 280$$

2) Using the equation for error propagation for division, total uncertainty is:

$$280\sqrt{\left(\frac{5}{28}\right)^2 + \left(\frac{0.01}{0.1}\right)^2} = 57 \text{ disintegrations}$$

3) The activity will then be 280 ÷ 5 minutes = 56 dpm and the total uncertainty will be 57 ÷ 5 minutes = 11 dpm. (Since the count time is considered to have trivial variance, this is assumed to be a constant.)

Referring to Table 6.9, a k value of ±1.96 represents a confidence interval equal to 95% about the mean of a normal distribution. Therefore, the 95% confidence interval would be 1.96 × 11 dpm = 22 dpm. The final result would be 56 ± 22 dpm.

6.9 Radon Measurements

There are three radon isotopes in nature: ^{222}Rn (radon) in the ^{238}U decay chain, ^{220}Rn (thoron) in the ^{232}Th chain, and ^{219}Rn (actinon) in the ^{235}U chain. ^{219}Rn is the least abundant of these three isotopes, and because of its short half-life of 4 seconds it has the least probability of emanating into the atmosphere before decaying. ^{220}Rn with a 55 second half-life is somewhat more mobile. ^{222}Rn with a 3.8 d half-life is capable of migrating through several decimeters of soil or building material and reaching the atmosphere. Therefore, in most situations, ^{222}Rn should be the predominant airborne radon isotope.

Many techniques have been developed over the years for measuring radon (Jenkins 1986) and radon progeny in air. In addition, considerable attention is given by EPA to measurement of radon and radon progeny in homes (EPA 1992d). Radon and radon progeny emit alpha and beta particles and gamma rays. Therefore, numerous techniques can and have been developed for measuring these radionuclides based on detecting alpha particles, beta particles, or gamma rays, independently or in some combination. It is even difficult to categorize the various techniques that are presently in use. This section contains an overview of information dealing with the measurement of radon and radon progeny. The information is focused on the measurement of ^{222}Rn, however the information may be adapted for the measurement of ^{219}Rn and ^{220}Rn.

Radon concentrations within a fixed structure can vary significantly from one section of the building to another and can fluctuate over time. If a home has a basement, for instance, it is usually expected that a higher radon concentration will be found there. Likewise, a relatively small increase in the relative pressure between the soil and the inside of a structure can cause a significant increase in the radon emanation rate from the soil into the structure. Many factors play a role in these variations, but from a practical standpoint it is only necessary to recognize that fluctuations are expected and that they should be accounted for. Long term measurement periods

are required to determine a true mean concentration inside a structure and to account for the fluctuations.

Two analytical end points are of interest when performing radon measurements. The first and most commonly used is radon concentration, which is stated in terms of activity per unit volume (Bq/m^3 or pCi/L). Although this terminology is consistent with most federal guidance values, it only infers the potential dose equivalent associated with radon. The second analytical end point is the radon progeny working level. Radon progeny usually attach very quickly to charged aerosols in the air following creation. The fraction that remains unattached is usually quite small (*i.e.*, 5-10%). Since most aerosol particles carry an electrical charge and are relatively massive ($\geq$ 0.1 μm), they are capable of attaching to the surfaces of the lung. Essentially all dose or risk from radon is associated with alpha decays from radon progeny attached to tissues of the respiratory system. If an investigator is interested in accurately determining the potential dose or risk associated with radon in the air of a room, the radon progeny concentration must be known.

Radon progeny concentrations are usually reported in units of working levels (WL), where one working level is equal to the potential alpha energy associated with the radon progeny in secular equilibrium with 100 pCi/L of radon. One working level is equivalent to 1.28×10^5 MeV/L of potential alpha energy. Given a known breathing rate and lung attachment probability, the expected mean lung dose from exposure to a known working level of radon progeny can be calculated.

Radon progeny are not usually found in secular equilibrium with radon indoors due to plating out of the charged aerosols onto walls, furniture, *etc.* The ratio of ^{222}Rn progeny activity to ^{222}Rn activity usually ranges from 0.2 to as high as 0.8 indoors (NCRP 1988). If only the ^{222}Rn concentration is measured and it is not practical to measure the progeny concentrations, then general practice is to assume a progeny to ^{222}Rn equilibrium ratio of 0.5 for indoor areas. This allows one to estimate the expected dose or risk associated with a given radon concentration.

In general, the following generic guidelines should be followed when performing radon measurements during site investigations:

- The radon measurement method used should be well understood and documented.
- Long term measurements are used to determine the true mean radon concentration.
- The impact of variable environmental conditions (*e.g.*, humidity, temperature, dust loading, and atmospheric pressure) on the measurement process should be accounted for when necessary. Consideration should be given to effects on both the air collection process and the counting system.

- The background response of the detection system should be accounted for.
- If the quantity of interest is the working level, then the radon progeny concentrations should be evaluated. If this is not practical, then the progeny activities can be estimated by assuming they are 50% of the measured radon activity (NCRP 1988).

For a general overview, a list of common radiation detectors with their usual applications during radon surveys is provided in Table 6.10. Descriptions and costs for specific equipment used for the measurement of radon are contained in Appendix H.

Table 6.10 Radiation Detectors with Applications to Radon Surveys

System	Description	Application	Remarks
Large area activated charcoal collector	A canister containing activated charcoal is twisted into the surface and left for 24 hours.	Short term radon flux measurements	The LLD is 0.007 Bq $m^{-2}s^{-1}$ (0.2 pCi $m^{-2}s^{-1}$).
Continuous radon monitor	Air pump and scintillation cell or ionization chamber.	Track the real time concentration of radon	Takes 1 to 4 hours for system to equilibrate before starting. The LLD is 0.004-0.04 Bq/L (0.1-1.0 pCi/L).
Activated charcoal adsorption	Activated charcoal is opened to the ambient air, then gamma counted on a gamma scintillator or in a liquid scintillation counter.	Measure radon concentration in indoor air	Detector is deployed for 2 to 7 days. The LLD is 0.007-0.04 Bq/L (0.2 to 1.0 pCi/L).
Electret ion chamber	This is a charged plastic vessel that can be opened for air to pass through.	Measure short-term or long-term radon concentration in indoor air	Must correct reading for gamma background concentration. Electret is sensitive to extremes of temperature and humidity. LLD is 0.007-0.02 Bq/L (0.2-0.5 pCi/L).
Alpha track detection	A small piece of special plastic or film inside a small container. Damage tracks from alpha particles are chemically etched and tracks counted.	Measure indoor or outdoor radon concentration in air	LLD is 0.04 Bq $L^{-1}d^{-1}$ (1 pCi $L^{-1}d^{-1}$).

The following provides a general overview of radon sampling and measurement concepts. The intent of this section is to provide an overview of common methods and terminology.

6.9.1 Direct Radon Measurements

Direct radon measurements are performed by gathering radon into a chamber and measuring the ionizations produced. A variety of methods have been developed, each making use of the same fundamental mechanics but employing different measurement processes. The first step is to get the radon into a chamber without collecting any radon progeny from the ambient air. A filter is normally used to capture charged aerosols while allowing the radon gas to pass through. Most passive monitors rely on diffusion of the ambient radon in the air into the chamber to establish an equilibrium between the concentrations of radon in the air and in the chamber. Active monitors use some type of air pump system for the air exchange method.

Once inside the chamber, the radon decays by alpha emission to form ^{218}Po which usually takes on a positive charge within thousandths of a second following formation. Some monitor types collect these ionic molecules and subsequently measure the alpha particles emitted by the radon progeny. Other monitor types, such as the electret ion chamber, measure the ionization produced by the decay of radon in the air within the chamber by directly collecting the ions produced inside the chamber. Simple systems measure the cumulative radon during the exposure period based on the total alpha decays that occur. More complicated systems actually measure the individual pulse height distributions of the alpha and/or beta radiation emissions and derive the radon plus progeny isotopic concentration in the air volume.

Care must be taken to accurately calibrate a system and to understand the effects of humidity, temperature, dust loading, and atmospheric pressure on the system. These conditions create a small adverse effect on some systems and a large influence on others.

6.9.1.1 Integrating Methods for Radon Measurement

With integrating methods, measurements are made over a period of days, weeks, or months and the device is subsequently read by an appropriate device for the detector media used. The most common detectors used are activated charcoal adsorbers, electret ion chamber (EIC), and alpha track plastics. Short term fluctuations are averaged out, thus making the measurement representative of average concentration. Results in the form of an average value provide no way to determine the fluctuations of the radon concentration over the measurement interval. Successive short term measurements can be used in place of single long term measurements to gain better insight into the time dependence of the radon concentration.

6.9.1.2 Continuous Methods for Radon Measurement

Devices that measure direct radon concentrations over successive time increments are generally called continuous radon monitors. These systems are more complex than integrating devices in that they measure the radon concentration and log the results to a data recording device on a real time basis. Continuous radon measurement devices normally allow the noble gas radon to pass

through a filter into a detection chamber where the radon decays and the radon and/or the resulting progeny are measured. The most common detectors used for real time measurements are ion chambers, solid state surface barrier detectors, and ZnS(Ag) scintillation detectors.

Continuous methods offer the advantage of providing successive, short-term results over long periods of time. This allows the investigator not only to determine the average radon concentration, but also to analyze the fluctuations in the values over time. More complicated systems are available that measure the relative humidity and temperature at the measurement location and log the values along with the radon concentrations to the data logging device. This allows the investigator to make adjustments, if necessary, to the resulting data prior to reporting the results.

6.9.2 Radon Progeny Measurements

Radon progeny measurements are performed by collecting charged aerosols onto filter paper and subsequently counting the filter for attached progeny. Some systems pump air through a filter and then automatically count the filter for alpha and/or beta emissions. An equivalent but more labor intensive method is to collect a sample using an air sampling pump and then count the filter in stand alone alpha and/or beta counting systems. The measurement system may make use of any number of different techniques ranging from full alpha and beta spectrometric analysis of the filters to simply counting the filter for total alpha and or beta emissions.

When performing total (gross) counting analyses, the assumption is usually made that the only radioisotopes in the air are due to ^{222}Rn and its progeny. This uncertainty, which is usually very small, can be essentially eliminated when performing manual sampling and analysis by performing a follow up measurement of the filter after the radon progeny have decayed to a negligible level. This value can then be used as a background value for the air. Of course, such a simple approach is only applicable when ^{222}Rn is the isotope of concern. For ^{219}Rn or ^{220}Rn, other methods would have to be used.

Time is a significant element in radon progeny measurements. Given any initial equilibrium condition for the progeny isotopes, an investigator must be able to correlate the sampling and measurement technique back to the true concentration values. When collecting radon progeny, the buildup of total activity on the filter increases asymptotically until the activity on the filter becomes constant. At this point, the decay rate of the progeny atoms on the filter is equal to the collection rate of progeny atoms. This is an important parameter to consider when designing a radon sampling procedure.

Note that the number of charged aerosol particles in the air can affect the results for radon progeny measurements. If the number of particles is few, as is possible when humidity is low and a room is very clean, then most of the progeny will not be attached and can plate out on room surfaces prior to reaching the sample filter. This is not a problem if the same conditions always

exist in the room, however the calculated dose would underestimate the dose that would be received in a higher humidity or dust concentration state with the same radon progeny concentration.

6.9.3 Radon Flux Measurements

Sometimes it is desirable to characterize the source of radon in terms of the rate at which radon is emanating from a surface—that is, soil, uranium mill tailings, or concrete. One method used for measuring radon flux is briefly described here.

The measurement of radon flux can be achieved by adsorption onto charcoal using a variety of methods such as a charcoal canister or a large area collector (*e.g.*, 25 cm PVC end cap). The collector is deployed by firmly twisting the end cap into the surface of the material to be measured. After 24 hours of exposure, the activated charcoal is removed and transferred to plastic containers. The amount of radon adsorbed on the activated charcoal is determined by gamma spectroscopy. Since the area of the surface is well defined and the deployment period is known, the radon flux (in units of Bq/m^2-s or pCi/m^2-s) can be calculated.

This method is reliable for measuring radon flux in normal environmental situations. However, care should be taken if an extremely large source of radon is measured with this method. The collection time should be chosen carefully to avoid saturating the canister with radon. If saturation is approached, the charcoal loses its ability to absorb radon and the collection rate decreases. Even transporting and handling of a canister that is saturated with radon can be a problem due to the dose rate from the gamma rays being emitted. One would rarely encounter a source of radon that is so large that this would become a problem; however, it should be recognized as a potential problem. Charcoal can also become saturated with water, which will affect the absorption of radon. This can occur in areas with high humidity.

An alternative method for making passive radon flux measurements has been developed recently using electret ionization chambers (EICs). EIC technology has been widely used for indoor radon measurements. The passive EIC procedure is similar to the procedures used with large area activated charcoal canisters. In order to provide the data for the background corrections, an additional passive monitor is located side by side on a radon impermeable membrane. These data are used to calculate the net radon flux. The Florida State Bureau of Radiation Protection has compared the results from measurements of several phosphogypsum flux beds using the charcoal canisters and EICs and has shown that the two methods give comparable results. The passive method seems to have overcome some of the limitations encountered in the use of charcoal. The measurement periods can be extended from hours to several days in order to obtain a better average, if needed. EIC flux measurements are not affected by environmental conditions such as temperature, humidity, and air flow. The measured sensitivities are comparable to the charcoal method but, unlike charcoal, EICs do not become saturated by humidity. Intermediate readings

can be made if needed.. In view of the low cost of the EIC reading/analyzing equipment, the cost per measurement can be as much as 50% lower than the charcoal method with additional savings in time.

6.10 Special Equipment

Various specialized systems have been developed which can be used during the performance of radiation surveys and site investigations. These range from specially designed quick radiation scanning systems to commercial global positioning systems (GPSs). The equipment may be designed to detect radiation directly, detect and locate materials associated with the contamination (*e.g.*, metal containers), or locate the position where a particular measurement is performed (*e.g.*, GPS). Because these specialized systems are continuously being modified and developed for site-specific applications, it is not possible to provide detailed descriptions of every system. The following sections provide examples of specialized equipment that have been applied to radiation surveys and site investigations.

6.10.1 Positioning Systems

As stated in Section 4.8.5, documenting the location of measurements is important for demonstrating the reproducibility of the results. There are a variety of positioning systems available that provide a range of accuracy and precision that can be evaluated during survey planning to determine their applicability to a particular site. These positioning systems can be used to establish a reproducible reference coordinate system or to locate individual measurements using an established reference coordinate system (*e.g.*, longitude and latitude).

6.10.1.1 Differential Global Positioning Systems

A variety of practical and versatile GPSs based on radio signals tracked from satellite beacons are available (e.g., Trimble™, Novatel™, Garmin™). These systems are generally used to aid in recording and retrieving location data with precision on the order of tens of meters. With a stationary base station and a separate moving locator, the system is deployed in the "differential global positioning system" (DGPS) mode. DGPSs can record and retrieve location data with a precision in the centimeter range.

DGPS can be used to provide position information on surface features in areas being surveyed, linking the survey results to previously published maps and aerial photographs. In addition, survey results may be positioned using the DGPS readings to accurately and precisely locate the results as well as the results of any subsequent analyses to these same maps or photographs. A process called waypointing uses the DGPS to locate specific points and allows the user to find

predetermined locations and set up gridded locations for measurements based on location data that are tied into local or state coordinate systems.

Limitations on the use of DGPS are related to the number of satellite beacons available to the system. When three or fewer satellites are available the accuracy and precision of the location data will be reduced. There are short periods of time (usually less than one hour even on the worst days) when a limited number of satellites are overhead in the continental United States. Satellites may also be blocked by excess tree cover or tall buildings. Distance between the moving locator and the stationary base station may be several kilometers or may be limited to line-of-sight. This limitation can be mitigated through the strategic use of repeater stations to re-transmit the signal between the moving locator and the base station.

6.10.1.2 Local Microwave and Sonar Positioning Systems

Local microwave or sonar beacons and receivers may provide useful location data in small areas and tree-covered locales. One example of a sonar-based system is the ultrasonic ranging and data system (USRADS). With a number of fixed beacons in place, a roving unit can be oriented and provide location data with similar accuracy and precision as the DGPS. If the beacons are located at known points, the resulting positions can be determined using simple calculations based on the known reference locations of the beacons.

The logistics of deploying the necessary number of beacons properly and the short range of the signals are the major limitations of the system. In addition, multipathing of signals within wooded areas can cause jumps in the positioning data.

6.10.2 Mobile Systems with Integrated Positioning Systems

In recent years, the advent of new technologies has introduced mobile sensor systems for acquiring data that include fully-integrated positioning systems. Portable and vehicle-based versions of these systems record survey data while moving over surfaces to be surveyed and simultaneously recording the location data from either a roving DGPS receiver or local microwave/sonar receiver. All measurement data are automatically stored and processed with the measurement location for later posting (see Section 8.2.2.2 for a discussion of posting plots) or for mapping the results. These systems are designed with a variety of detectors for different applications. For example, alpha or beta detectors have been mounted on a robot a fixed distance over a smooth surface. The robot moves at a predetermined speed over the surface to provide scanning results, and also records individual direct measurements at predetermined intervals. This type of system not only provides the necessary measurement data, but also reduces the uncertainty associated with human factors. Other systems are equipped with several types of radiation detectors, magnetometers, electromagnetic sensors, or various combinations of multiple sensors. The limitations of each system should be evaluated on a site-specific basis to determine if the

positioning system, the detector, the transport system, or some combination based on site-specific characteristics will represent the limits of the system.

6.10.3 Radar, Magnetometer, and Electromagnetic Sensors

The number of sensors and sensor systems applicable to the detection and location of buried waste have increased in use and reliability in recent years. These systems are typically applicable to scoping and characterization surveys where the identification of subsurface contamination is a primary concern. However, the results of these surveys may be used during final status survey planning to demonstrate that subsurface contamination is not a concern for a particular site or survey unit. Some of the major technologies are briefly described in the following sections.

6.10.3.1 Ground Penetrating Radar

For most sites, ground penetrating radar (GPR) is the only instrument capable of collecting images of buried objects *in situ*, as compared to magnetometers (Section 6.10.3.2) and electromagnetic sensors (Section 6.10.3.3) which detect the strength of signals as measured at the ground surface. Additionally, GPR is unique in its ability to detect both metallic and non-metallic (*e.g.*, plastic, glass) containers.

Subsurface radar detection systems have been the focus of study for locating and identifying buried or submerged objects that otherwise could not be detected. There are two major categories of radar signals: 1) time domain, and 2) frequency domain. Time-domain radar uses short impulses of radar-frequency energy directed into the ground being investigated. Reflections of this energy, based on changes in dielectric properties, are then received by the radar. Frequency-domain radar, on the other hand, uses a continuous transmission where the frequency of the transmission can be varied either stepwise or continuously. The changes in the frequency characteristics due to effects from the ground are recorded. Signal processing, in both cases, converts this signal to represent the location of radar reflectors against the travel time of the return signal. Greater travel time corresponds to a greater distance beneath the surface. Table 6.11 lists the typical penetration depth for various geologic materials (fresh water is included as a baseline for comparison).

Examples of existing GPR technologies currently being applied to subsurface investigations include:

- narrow-band radar
- ultra-wideband radar
- synthetic aperture radar
- frequency modulated continuous radar
- polarized radar waves

Table 6.11 Typical Radar Penetration Depths for Various Geologic Materials

Material	Penetration Depth m (ft)
Fresh Water	100 (330)
Sand (desert)	5 (16)
Sandy Soil	3 (10)
Loam Soil	3 (10)
Clay Soil	2 (6)
Salt Flats (dry)	1 (3)
Coal	20 (66)
Rocks	20 (66)
Walls	0.3 (1)

The major limitation to GPR is the difficulty in interpreting the data, which is often provided in the form of hazy, "waterfall-patterned" data images requiring an experienced professional to interpret. Also, GPR can vary depending on the soil type as shown in Table 6.10. Highly conductive clay soils often absorb a large amount of the radar energy, and may even reflect the energy. GPR can be deployed using ground-based or airborne systems.

6.10.3.2 Magnetometers

Although contaminated soil and most radioactive waste possess no ferromagnetic properties, the containers commonly used to hold radioactive waste (*e.g.*, 55-gallon drums) are made from steel. These containers possess significant magnetic susceptibility making the containers detectable using magnetometry.

Magnetometers sense the pervasive magnetic field of the Earth. This field, when encountering an object with magnetic susceptibility, induces a secondary magnetic field in that object. This secondary field creates an increase or decrease in Earth's ambient magnetic field. Magnetometers measure these changes in the expected strength of the ambient magnetic field. Some magnetometers, called "vector magnetometers," can sense the direction as well as the magnitude of these changes. However, for subsurface investigations only the magnitude of the changes are used.

The ambient magnetic field on Earth averages 55,000 gamma in strength. The variations caused by the secondary magnetic fields typically range from 10 to 1,000 gamma, and average around 100 gamma. Most magnetometers currently in use have a sensitivity in the 0.1 to 0.01 gamma range and are capable of detecting these secondary fields.

An alternate magnetometer survey can be performed using two magnetometers in a gradiometric configuration. This means that the first magnetometer is placed at the ground surface, while the second is mounted approximately 0.5 meters above the first. Data is recorded from both sensors and compared. When the readings from both detectors are nearly the same, it implies that there is no significant disturbance in the Earth's ambient magnetic field or that such disturbances are broad and far away from the gradiometer. When a secondary magnetic field is induced in an object, it affects one sensor more strongly than the other, producing a difference in the readings from the two magnetometers. This approach is similar to the use of a guard detector in anti-coincidence mode in a low-background gas-flow proportional counter in a laboratory (see Appendix H for a description of gas-flow proportional counters). The gradiometric configuration filters out the Earth's ambient magnetic field, large scale variations, and objects located far from the sensor to measure the effects of nearby objects, all without additional data processing.

Fifty-five gallon drums buried 5 to 7 meters below the surface may be detectable using a magnetometer. At many sites, multiple drums have been buried in trenches or pits and detection is straightforward. A single operator carrying a magnetometer with the necessary electronics in a backpack can cover large areas in a relatively small amount of time.

The limitations on the system are related to the size of the objects and their depth below the surface. Objects that are too small or buried too deep will not provide a secondary magnetic field that can be detected at the ground surface.

6.10.3.3 Electromagnetic Sensors

Electromagnetic sensors emit an electromagnetic wave, in either a pulsed or continuous wave mode, and then receive the result of that transmission. The result of the transmission is two signals; quadrature and in-phase. As the wave passes through some material other than air, it is slowed down by a resistive medium or sped up by a conductor through dielectric effects. This produces the quadrature signal. If the electromagnetic wave encounters a highly conductive object it induces a magnetic field in the object. This induced electromagnetic field returns to the sensor as a reflection of the original electromagnetic wave and forms the in-phase signal.

The in-phase signal is indicative of the presence, size, and conductivity of nearby objects (*e.g.*, 55-gallon drums), while the quadrature signal is a measure of the dielectric properties of the nearby objects such as soil. This means that electromagnetic sensors can detect all metallic objects (including steel, brass, and aluminum), such as the metal in waste containers, and also sample the soil for changes in properties, such as those caused by leaks of contaminants.

Depths of interest are largely determined by the spacing between the coil used to transmit the primary electromagnetic wave, and the receiver used to receive that transmission. The rule of thumb is that the depth of interest is on the order of the distance between the transmitter and the receiver. A system designed with the transmitter and receiver placed tens of meters apart can detect signals from tens of meters below the surface. A system with the transmitter and receiver collocated can only detect signals from depths on the order of the size of the coil, which is typically about one meter. The limitations of electromagnetic sensors include a lack of clearly defined signals, and decreasing resolution of the signal as the distance below the surface increases.

6.10.4 Aerial Radiological Surveys

Low–altitude aerial radiological surveys are designed to encompass large areas and may be useful in:

- providing data to assist in the identification of radioactive contaminants and their corresponding concentrations and spatial distributions
- characterizing the nature, extent, and impact of contamination

The measurement sensitivity and data processing procedures provide total area coverage and a detailed definition of the extent of gamma-producing isotopes for a specific area. The gamma radiation spectral data are processed to provide a qualitative and quantitative analysis of the radionuclides in the survey area. Helicopter flights establish a grid pattern (*e.g.*, east–west) of parallel lines approximately 61 m (200 ft) above the ground surface.

The survey consists of airborne measurements of natural and man–made gamma radiation from the terrain surface. These measurements allow for the determination of terrestrial spatial distribution of isotopic concentrations and equivalent gamma exposure rates (*e.g.*, ^{60}Co, ^{234m}Pa, and ^{137}Cs). The results are reported as isopleths for the isotopes and are usually superimposed on scale maps of the area.

7 SAMPLING AND PREPARATION FOR LABORATORY MEASUREMENTS

7.1 Introduction

There are three methods for collecting radiation data while performing a survey. A direct measurement is obtained by placing the detector near or against the surface or in the media being surveyed and reading the radioactivity level directly. Scanning is an evaluation technique performed by moving a portable radiation detection instrument at a constant speed and distance above the surface to semi-quantitatively detect elevated areas of radiation. These measurement techniques are discussed in Chapter 6. Sampling is the process of collecting a portion of an environmental medium as representative of the locally remaining medium. The collected portion of the medium is then analyzed to determine the radionuclide concentration. This chapter discusses issues involved in collecting and preparing samples in the field for analysis, and in evaluating the results of these analyses. In addition, a general discussion on laboratory sample preparation and analysis is provided to assist in communications with the laboratory during survey planning.

Samples should be collected and analyzed by qualified individuals using the appropriate equipment and procedures. This manual assumes that the samples taken during the survey will be submitted to a qualified laboratory for analysis. The laboratory should have written procedures that document its analytical capabilities for the radionuclides of interest and a Quality Assurance/Quality Control (QA/QC) program that documents the compliance of the analytical process with established criteria. The method used to assay for the radionuclides of concern should be recognized as a factor affecting analysis time.

Commonly used radiation detection and measuring equipment for radiological survey field applications is described in Chapter 6 and Appendix H. Many of these equipment types are also used for laboratory analyses, usually under more controlled conditions that provide for lower detection limits and greater delineation between radionuclides. Laboratory methods often involve combinations of both chemical and instrument techniques to quantify the low levels expected in the samples. This chapter provides guidance to assist the MARSSIM user in selecting appropriate procedures for collecting and handling samples for laboratory analysis. More detailed information is available in documents listed in the reference section of this manual.

7.2 Data Quality Objectives

The survey design is developed and documented using the Data Quality Objectives (DQO) Process (see Appendix D). The third step of the DQO Process involves identifying the data needs for a survey. One decision that can be made at this step is the selection of direct measurements

for performing a survey or deciding that sampling methods followed by laboratory analysis are necessary.

7.2.1 Identifying Data Needs

The decision maker and the survey planning team need to identify the data needs for the survey being performed, including the:

- type of samples to be collected or measurements to be performed (Chapter 5)
- radionuclide(s) of interest (Section 4.3)
- number of samples to be collected (Section 5.5.2)
- type and frequency of field QC samples to be collected (Section 4.9)
- amount of material to be collected for each sample (Section 4.7.3 and Section 7.5)
- sampling locations and frequencies (Section 5.5.2)
- standard operating procedures (SOPs) to be followed or developed (Chapter 7)
- analytical bias and precision (*e.g.*, quantitative or qualitative) (Appendix N)
- target detection limits for each radionuclide of interest (Section 6.4 and Table 7.2)
- cost of the methods being evaluated (cost per analysis as well as total cost) (Appendix H)
- necessary turnaround time
- sample preservation and shipping requirements (Section 7.6 and Section 7.9)
- specific background for the radionuclide(s) of interest (Section 4.5)
- derived concentration guideline level (DCGL) for each radionuclide of interest (Section 4.3)
- measurement documentation requirements (Section 9.4.2.2)
- sample tracking requirements (Section 7.8)

Some of this information will be supplied by subsequent steps in the DQO process, and several iterations of the process may be needed to identify all of the data needs. Consulting with a radiochemist or health physicist may be necessary to properly evaluate the information before deciding between direct measurements or sampling methods to perform the survey. Surveys may require data from all three collection methods (*i.e.*, sample analysis, direct measurements, and scans) in order to demonstrate compliance with the regulation.

7.2.2 Data Quality Indicators

The data quality indicators identified as DQOs in Section 2.3.1 and described in Appendix N, Section N.6, should be considered when selecting a measurement method (*i.e.*, scanning, direct measurement, sampling) or an analytical technique (*e.g.*, radionuclide-specific analytical procedure). In some instances, the data quality indicator requirements will help in the selection of an analytical technique. In other cases, the analytical requirements will assist in the selection of appropriate levels for the data quality indicators.

7.2.2.1 Precision

Precision is a measure of agreement among replicate measurements of the same property under prescribed similar conditions (ASQC 1995). Precision is determined quantitatively based on the results of replicate measurements (equations are provided in EPA 1990). The number of replicate analyses needed to determine a specified level of precision for a project is discussed in Section 4.9. There are several types of replicate analyses available to determine the level of precision, and these replicates are typically distinguished by the point in the sample collection and analysis process where the sample is divided. Determining precision by replicating measurements with results at or near the detection limit of the measurement system is not recommended because the measurement uncertainty is usually greater than the desired level of precision.

- Collocated Samples. Collocated samples are samples collected adjacent to the routine field sample to determine local variability of the radionuclide concentration. Typically, collocated samples are collected about one-half to three feet away from the selected sample location. Analytical results from collocated samples can be used to assess site variation, but only in the immediate sampling area. Collocated samples should not be used to assess variability across a site and are not recommended for assessing error (EPA 1991g). Collocated samples can be non-blind, single-blind, or double-blind.

- Field Replicates. Field replicates are samples obtained from one location, homogenized, divided into separate containers and treated as separate samples throughout the remaining sample handling and analytical processes. These samples are used to assess error associated with sample heterogeneity, sample methodology and analytical procedures. Field replicates are used when determining total error for critical samples with contamination concentrations near the action level. For statistical analysis to be valid in such a case, a minimum of eight replicate samples would be required (EPA 1991g). Field replicates (or field split samples) can be non-blind, single-blind, or double-blind and are recommended for determining the level of precision for a radiation survey or site investigation.

- Analytical Laboratory Replicate. An analytical laboratory replicate is a subsample of a routine sample that is homogenized, divided into separate containers, and analyzed using the same analytical method. It is used to determine method precision, but because it is a non-blind sample, or known to the analyst, it can only be used by the analyst as an internal control tool and not as an unbiased estimate of analytical precision (EPA 1990).

- Laboratory Instrument Replicate. A laboratory instrument replicate is the repeated measurement of a sample that has been prepared for counting (*i.e.*, laboratory sample preparation and radiochemical procedures have been completed). It is used to determine

precision for the instrument (repeated measurements using same instrument) and the instrument calibration (repeated measurements using different instruments, such as two different germanium detectors with multichannel analyzers). A laboratory instrument replicate is generally performed as part of the laboratory QC program and is a non-blind sample. It is typically used as an internal control tool and not as an unbiased estimate of analytical precision.

7.2.2.2 Bias

Bias is the systematic or persistent distortion of a measurement process that causes error in one direction (ASQC 1995). Bias is determined quantitatively based on the analysis of samples with a known concentration. There are several types of samples with known concentrations. QC samples used to determine bias should be included as early in the analytical process as possible.

- Reference Material. A material or substance one or more of whose property values are sufficiently homogeneous and well established to be used for the calibration of an apparatus, the assessment of a measurement method, or for assigning values to materials (ISO 1993). A certified reference material is reference material for which each certified property value is accompanied by an uncertainty at a stated level of confidence. Radioactive reference materials may be available for certain radionuclides in soil (*e.g.*, uranium in soil), but reference building materials may not be available. Because reference materials are prepared and homogenized as part of the certification process, they are rarely available as double-blind samples. When appropriate reference materials are available (*i.e.*, proper matrix, proper radionuclide, proper concentration range), they are recommended for use in determining the overall bias for a measurement system.

- Performance Evaluation (PE) Samples. PE sample are samples that evaluate the overall bias of the analytical laboratory and detect any error in the analytical method used. These samples are usually prepared by a third party, using a quantity of analyte(s) which is known to the preparer but unknown to the laboratory, and always undergo certification analysis. The analyte(s) used to prepare the PE sample is the same as the analyte(s) of interest. Laboratory procedural error is evaluated by the percentage of analyte identified in the PE sample (EPA 1991g). PE samples are recommended for use in determining overall bias for a measurement system when appropriate reference material are not available. PE samples are equivalent to matrix spikes prepared by a third party that undergo certification analysis and can be non-blind, single-blind, or double-blind.

- Matrix Spike Samples. Matrix spike samples are environmental samples that are spiked in the laboratory with a known concentration of a target analyte(s) to verify percent recoveries. They are used primarily to check sample matrix interferences but can also be used to monitor laboratory performance. However, a data set of at least three or more

results is necessary to distinguish between laboratory performance and matrix interference (EPA 1991g). Matrix Spike samples are often replicated to monitor method performance and evaluate error due to laboratory bias and precision (when four or more pairs are analyzed). These replicates are often collectively referred to as a matrix spike/matrix spike duplicate (MS/MSD).

There are several additional terms applied to samples prepared by adding a known amount of the radionuclide of interest to the sample. The majority of these samples are designed to isolate individual sources of bias within a measurement system by preparing pre- and post-operation spikes. For example, the bias from the digestion phase of the measurement system can be determined by comparing the result from a pre-digest spike to the result from a post-digest spike.

There are also several types of samples used to estimate bias caused by contamination.

- Background Sample. A background sample is a sample collected upgradient of the area of potential contamination (either onsite or offsite) where there is little or no chance of migration of the contaminants of concern (EPA 1991g). Background samples are collected from the background reference area (Section 4.5), determine the natural composition and variability of the soil (especially important in areas with high concentrations of naturally occurring radionuclides), and are considered "clean" samples. They provide a basis for comparison of contaminant concentration levels with samples collected from the survey unit when the statistical tests described in Chapter 8 are performed.

- Field Blanks. Field blanks are samples prepared in the field using certified clean sand or soil and then submitted to the laboratory for analysis (EPA 1991g). A field blank is used to evaluate contamination error associated with sampling methodology and laboratory procedures. It also provides information about contaminants that may be introduced during sample collection, storage, and, transport. Field blanks are recommended for determining bias resulting from contamination for a radiation survey or site investigation.

- Method Blank. A method blank is an analytical control sample used to demonstrate that reported analytical results are not the result of laboratory contamination (ATSDR 1992). It contains distilled or deionized water and reagents, and is carried through the entire analytical procedure (laboratory sample preparation, digestion, and analysis). The method blank is also referred to as a reagent blank. The method blank is generally used as an internal control tool by the laboratory because it is a non-blind sample.

7.2.2.3 Representativeness

Representativeness is a measure of the degree to which data accurately and precisely represent a characteristic of a population parameter at a sampling point (ASQC 1995). Representativeness is a qualitative term that is reflected in the survey design through the selection of a measurement method (*e.g.*, direct measurement or sampling) and the size of a sample collected for analysis.

Sample collection and analysis is typically less representative of true radionuclide concentrations at a specific measurement location than performing a direct measurement. This is caused by the additional steps required in collecting and analyzing samples, such as sample collection, field sample preparation, laboratory sample preparation, and radiochemical analysis. However, direct measurement techniques with acceptable detection limits are not always available. When sampling is required as part of a survey design, it is critical that the sample collection procedures consider representativeness. The location of the sample is determined in Section 5.5.2.5, but the size and content of the sample are usually determined as the sample is collected. Sample size and content are discussed in Section 4.7.3 and Section 7.5. Sample collection procedures also need to consider the development of the DCGLs when determining the representativeness of the samples.

7.2.2.4 Comparability

Comparability is a qualitative term that expresses the confidence that two data sets can contribute to a common analysis and interpolation. Generally, comparability is provided by using the same measurement system for all analyses of a specific radionuclide. In many cases, equivalent procedures used within a measurement system are acceptable. For example, using a liquid-liquid extraction purification step to determine the concentration of ^{238}Pu using alpha spectrometry may be equivalent to using an ion-exchange column purification step. However, using a gross alpha measurement on a gas proportional counting system would not be considered equivalent. Comparability is usually not an issue except in cases where historical data have been collected and are being compared to current analytical results, or when multiple laboratories are used to provide results as part of a single survey design.

7.2.2.5 Completeness

Completeness is a measure of the amount of valid data obtained from the measurement system, expressed as a percentage of the number of valid measurements that should have been collected. Completeness is of greater concern for laboratory analyses than for direct measurements because the consequences of incomplete data often require the collection of additional samples. Direct measurements can usually be repeated fairly easily. The collection of additional samples generally requires a remobilization of sample collection personnel which can be expensive. Conditions at the site may have changed making it difficult or impossible to collect representative and

comparable samples without repeating the entire survey. On the other hand, if it is simply an analytical problem and sufficient sample was originally collected, the analysis can be repeated using archived sample material. Samples collected on a grid to locate areas of elevated activity are also a concern for completeness. If one sample analysis is not valid, the entire survey design for locating areas of elevated activity may be invalidated.

7.2.2.6 Other Data Quality Indicators

Several additional data quality indicators that influence the final status survey design are identified as DQOs in Section 2.3.1. Many of these (*e.g.*, selection and classification of survey units, decision error rates, variability in the contaminant concentration, lower bound of the gray region) are used to determine the number of measurements and are discussed in detail in Section 5.5. The method detection limit is directly related to the selection of a measurement method and a radionuclide-specific analytical technique.

Analytical methods should be capable of measuring levels below the established DCGLs, detection limits of 10-50% of the DCGL should be the target (see Section 6.7). Cost, time, best available technology, or other constraints may create situations where the above stated sensitivities are deemed impracticable. Under these circumstances, higher detection sensitivities may be acceptable. Although laboratories will state detection limits, these sensitivities are usually based on ideal or optimistic situations and may not be achievable under actual measurement conditions. Detection limits are subject to variation from sample to sample, instrument to instrument, and procedure to procedure, depending on sample size, geometry, background, instrument efficiency, chemical recovery, abundance of the radiations being measured, counting time, self-absorption in the prepared sample, and interferences from radionuclides or other materials present in the sample. The detection limit that is achievable in practice should not exceed the DCGL.

7.3 Communications with the Laboratory

Laboratory analyses of samples are generally performed by personnel not directly involved in the collection of the samples being analyzed. Samples are typically collected by one group working in the field, and analyzed by a second group located in a laboratory. This separation of tasks can potentially lead to problems based on the lack of communication between the two groups. For this reason, communications between the Project Manager, field personnel, and laboratory personnel are vital to ensuring the success of a project.

7.3.1 Communications During Survey Planning

The radioanalytical laboratory is a valuable resource during survey planning. Information on available analytical techniques, analytical bias and precision, method detection limits, analytical costs, and turnaround times can easily be provided by the laboratory. All of this information is used to make the decision to perform direct measurements or collect samples for laboratory measurements. Additional information, such as required sample size/volume, type of sample container, preservative requirements, and shipping requirements, including the availability of the laboratory for receipt of samples on weekends or holidays, can be obtained and factored into the survey plan.

Involving the radioanalytical laboratory during survey planning also provides the laboratory with site-specific information about the project. Information on the radionuclides of interest, possible chemical and physical form of the contamination, and mechanism for release of the contamination to the environment is used to modify or develop the analytical method for site-specific conditions if required. The laboratory should also be provided with the site-specific action levels (i.e., DCGLs, investigation levels) early in the survey planning process.

In some cases, it is not practical to select a radioanalytical laboratory early in the survey process to participate in the survey planning activities. For example, Federal procurement procedures require that a statement of work (SOW) identifying the tasks to be performed by the laboratory be developed prior to selecting a laboratory. Unfortunately, the details of the tasks for the laboratory to perform are developed during survey planning. This means that the information provided by the laboratory and used during survey planning will be obtained from another source, usually a radiochemist or health physicist trained in radiochemistry. The uncertainty associated with this information and subsequent decisions made based on this information increases. This may lead to increased costs caused by specifying an unnecessarily expensive analytical method in the SOW or repeated sampling and analysis of samples that did not meet the target detection limits because the specified analytical method was not sensitive enough. In addition, unnecessary or inappropriate analytical methods may be selected by the laboratory because site-specific information concerning the samples was not provided.

The laboratory should be consulted when planning the schedule for the survey to insure that the expected turnaround times can be met based on the projected laboratory workload.

7.3.2 Communications Before and During Sample Collection

In most situations, the sample collection and shipping containers are supplied by the laboratory; therefore, the laboratory should be notified well in advance of the sampling trip so that these items will be available to the sampling team during the survey.

The main purpose of communications with the laboratory during sample collection is to inform the laboratory of modifications to the survey design specified in the planning documents (*e.g.*, QAPP and SOPs). The laboratory should have a copy of the survey design in their possession prior to samples being collected.

Modifications to the survey design are often minor deviations from the SOPs caused by site-specific conditions and usually affect a small number of samples. For example, a rock outcropping covered by a thin layer of soil may restrict the depth of the surface soil sample to 5 cm (2 in.) instead of the 10 cm (4 in.) specified in the SOP. The mass of the samples collected from this area of the site is one-half the expected sample mass, and the laboratory needs to be informed of this deviation from the SOP.

In other situations, there may be an extensive modification to the number or types of samples collected at the site that will affect the analytical methods, detection capabilities, analytical costs, or even the assumptions used to develop the DCGL. For example, a large portion of the site may have been converted to a parking lot. A large pile of material that may represent the former surface soil will be sampled as well as soil collected from beneath the parking lot surface. The number of samples to be analyzed has doubled compared to the original SOW.

If the expected timing of receipt of samples at the laboratory changes due to sample collection schedule deviations, the laboratory should be notified. Most laboratories require prior notification for samples to be received on weekends.

7.3.3 Communications During Sample Analysis

The laboratory should communicate with the Project Manager and field personnel during sample analysis. The laboratory should provide a list of missing or damaged samples as soon after the samples are received as practical. This allows the Project Manager to determine if resampling is required to replace the missing or damaged samples. The Project Manager may also request notification from the laboratory when samples are spilled or lost during analysis. Preliminary reports of analytical results may be useful to help direct sampling activities and provide early indications of whether the survey objectives defined by the DQOs are being met. However, if preliminary results have not been verified or validated, their usefulness is limited.

7.3.4 Communications Following Sample Analysis

Following sample analysis, the laboratory will provide documentation of the analytical results as specified in the survey design. Laboratory personnel should be available to assist with data verification and validation.

7.4 Selecting a Radioanalytical Laboratory

Once the decision to perform sampling activities is made, the next step is to select the analytical methods and determine the data needs for these methods. It is advisable to select a radiochemical laboratory early in the survey planning process in order that it may be consulted on the analytical methodology[1] and the sampling activities. In addition, mobile laboratories can provide on-site analytical capability. Obtaining laboratory or other services may involve a specific procurement process. Federal procurement procedures may require additional considerations beyond the method described here.

The procurement of laboratory services usually starts with the development of a request for proposal that includes a statement-of-work describing the analytical services to be procured. The careful preparation of the statement-of-work is essential to the selection of a laboratory capable of performing the required services in a technically competent and timely manner.

The technical proposals received in response to the procurement request for proposal must be reviewed by personal familiar with radioanalytical laboratory operations in order to select the most qualified offerer. For complicated sites with a large number of laboratory analyses, it is recommended that a portion of this evaluation take the form of a pre-award audit. The provision for this audit must be in the request for proposal. The results of this audit provide a written record of the decision to use a specific laboratory. Smaller sites or facilities may decide that a review of the laboratory's qualifications is sufficient for the evaluation.

There are six criteria that should be reviewed during this evaluation:

- Does the laboratory possess the appropriate well-documented procedures, instrumentation, and trained personnel to perform the necessary analyses? Necessary analyses are defined by the data needs (radionuclide(s) of interest and target detection limits) identified by the DQO process.

- Is the laboratory experienced in performing the same or similar analyses?

- Does the laboratory have satisfactory performance evaluation results from formal monitoring or accreditation programs? The laboratory should be able to provide a summary of QA audits and proof of participation in interlaboratory cross-check programs. Equipment calibrations should be performed using National Institute of Standards and Technology (NIST) traceable reference radionuclide standards whenever possible.

[1] The laboratory provides information on personnel, capabilities, and current workload that are necessary inputs to the decision-making process.

- Is there an adequate capacity to perform all analyses within the desired timeframe? This criterion considers whether or not the laboratory possesses a radioactive materials handling license or permit for the samples to be analyzed. Very large survey designs may indicate that more than one analytical laboratory is necessary to meet the survey objectives.[2]

- Does the laboratory provide an internal quality control review of all generated data that is independent of the data generators?

- Are there adequate protocols for method performance documentation and sample security?

Providers of radioanalytical services should have an active and fully documented QA program in place.[3] This program should comply with the objectives determined by the DQO process in Section 2.3. The QA program should include:

- laboratory organizational structure
- personnel qualifications
- written standard operating procedures and instructions
- inter- and intralaboratory performance analyses
- design control to define the flow of samples through the laboratory
- a corrective action plan
- an internal audit program

Chain-of-Custody requirements and numbers of samples are also specified. The analytical procedures as well as the documentation and reporting requirements should be specified and agreed upon. These topics are discussed in detail in the following sections of this chapter.

7.5 Sampling

This section provides guidance on developing appropriate sample collection procedures for surveys designed to demonstrate compliance with a dose- or risk-based regulation. Sample collection procedures are concerned mainly with ensuring that a sample is representative of the sample media, is large enough to provide sufficient material to achieve the desired detection limit, and is consistent with assumptions used to develop the conceptual site model and the DCGLs. Additional considerations for sample collection activities are discussed in Section 4.7.3.

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[2] If several laboratories are performing analyses as part of the survey, the analytical methods used to perform the analyses should be similar to ensure comparability of results (see Appendix N, Section N.6.5).

[3] The QA program is typically documented in one or more documents such as a Quality Management Plan, Quality Assurance Manual, or Quality Assurance Project Plan.

The presence of radioactive and hazardous chemical wastes (mixed wastes) at a site can influence the survey design. The external exposure rates or radioactivity concentration of a specific sample may limit the time that workers will be permitted to remain in intimate contact with the samples, or may dictate that smaller samples be taken and special holding areas be provided for collected samples prior to shipment. These special handling considerations may conflict with the size specifications for the analytical method, normal sampling procedures, or equipment. There is a potential for biasing sampling programs by selecting samples that can be safely handled or legally shipped to support laboratories. Because final status surveys are performed to demonstrate that a site can be safely released, issues associated with high levels of radioactivity are not expected to be a concern.

7.5.1 Surface Soil

The purpose of surface soil sampling is to collect samples that accurately and precisely represent the radionuclides and their concentrations at the location being sampled. In order to do this and plan for sampling, a decision must be made as to the survey design. The selection of a survey design is based on the Historical Site Assessment, results from preliminary surveys (*i.e.*, scoping characterization, remedial action support), and the objectives of the survey developed using the Data Quality Objectives (DQO) Process. The selection between judgmental, random, and systematic survey designs is discussed in Section 5.5.3.

7.5.1.1 Sample Volume

The volume of soil collected should be specified in the sample collection procedure. In general, large volumes of soil are more representative than small volumes of soil. In addition, large samples provide sufficient sample to ensure that required detection limits can be achieved and that sample reanalysis can be done if there is a problem. However, large samples may cause problems with shipping, storage, and disposal. All of these issues should be discussed with the sample collection team and the analytical laboratory during development of sample collection procedures. In general, surface soil samples range in size from 100 g up to several kilograms.

The sample collection procedure should also make clear if it is more important to meet the volume requirement of the survey design or the surface area the sample represents. Constant volume is related to comparability of the results while surface area is more closely related to the representativeness of the results. Maintaining a constant surface area and depth for samples collected for a particular survey can eliminate problems associated with different depth profiles. The actual surface area included as part of the sample may be important for estimating the probability of locating areas of elevated concentration.

7.5.1.2 Sample Content

The material present in the field at the sample location may or may not provide a representative sample. Vegetative cover, soil particle size distribution, inaccessibility, or lack of sample material are examples of problems that may be identified during sample collection. All deviations from the survey design as documented in the Standard Operating Procedures (SOPs) should be recorded as part of the field sample documentation.

Sample content is generally defined by the assumptions used to develop the conceptual site model and the DCGLs. A typical agricultural scenario assumes that the top few centimeters of soil are available for resuspension in air, that the top 15 cm (6 in.) are homogenized by agricultural activities (*e.g.*, plowing), that roots can extend down several meters to obtain water and nutrients depending on the plant, and that external exposure is based on an assumed thickness of contaminated soil (usually at the surface). Depending on the dominant exposure pathways for each radionuclide, this can result in a complicated set of instructions for collecting representative samples. This situation can be further complicated by the fact that the site is not currently being used for agricultural purposes. For this situation it is necessary to look at the analytical results from the preliminary surveys (*i.e.*, scoping, characterization, remedial action support) to determine the expected depth of contamination.

In most situations the vegetative cover is not considered part of the surface soil sample and is removed in the field. For agricultural scenarios where external exposure is not the primary concern, soil particles greater than 2 mm (0.08 in.) are generally not considered as part of the sample (EPA 1990). Foreign material (*e.g.*, plant roots, glass, metal, or concrete) is also generally not considered part of the sample, but should be reviewed on a site-specific basis. It is important that the sample collection procedure clearly indicate what is and what is not considered part of the sample.

7.5.1.3 Sampling Equipment

The selection of proper sampling equipment is important to ensure that samples are collected effectively and efficiently. Sampling equipment generally consists of a tool to collect the sample and a container to place the collected sample in. Sample tracking begins as soon as the sample is collected, so it may be necessary to consider security of collected samples required by the objectives of the survey.

Sampling tools are selected based on the type of soil, sample depth, number of samples required, and training of available personnel. The selection of a sampling tool may also be based on the expected use of the results. For example, if a soil sample is collected to verify the depth profile used to develop the calibration for *in situ* gamma spectrometry, it is important to preserve the soil core. Table 7.1 lists several examples of tools used for collecting soil samples, situations where they are applicable, and some advantages and disadvantages involved in their use.

Table 7.1 Soil Sampling Equipment*

Equipment	Application	Advantages/Disadvantages
Tier	Soft surface soil	Inexpensive; easy to use and decontaminate; difficult to use in stone or dry soil.
Scoop or trowel	Soft surface soil	Inexpensive; easy to use and decontaminate; trowels with painted surfaces should be avoided
Bulb Planter	Soft Soil, 0-15 cm (0-6 in.)	Easy to use and decontaminate: uniform diameter and sample volume; preserves soil core; limited depth capability; can be difficult to decontaminate
Soil Coring Device	Soft soil, 0-60 cm (0-24 in.)	Relatively easy to use; preserves soil core; limited depth capability; can be difficult to decontaminate
Thin-wall tube sampler	Soft soil, 0-3 m (0-10 ft)	easy to use; preserves soil core; easy to decontaminate; can be difficult to remove cores
Split spoon sampler	Soil, to bedrock	Excellent depth range; preserves soil core; useful for hard soils; often used in conjunction with drill rig for obtaining deep cores
Shelby tube sampler	Soft soil, to bedrock	Excellent depth range; preserves soil core; tube may be used for shipping core to lab.; may be used in conjunction with drill rig for obtaining deep cores
Bucket auger	Soft soil, 7.5 cm - 3 m (3 in. - 10 ft)	Easy to use; good depth range; uniform diameter and sample volume; may disrupt and mix soil horizons greater than 15 cm
Hand -operated power auger	Soil, 15 cm - 4.5 m (6 in. -15 ft)	Good depth range; generally used in conjunction with bucket auger; destroys soil core; requires two or more operators; can be difficult to decontaminate

* Reproduced from EPA 1991g

Sample containers are generally not a major concern for collecting surface soil samples. Polyethylene bottles with screw caps and wide mouths are recommended. These containers are fairly economical, provide easy access for adding and removing samples, and resist chemicals, breaking, and temperature extremes. Glass containers are also acceptable, but they are fragile and tend to break during shipment. Metal containers are sometimes used, but sealing the container can present a problem and corrosion can be an issue if the samples are stored for a significant length of time.

7.5.2 Building Surfaces

Because building surfaces tend to be relatively smooth and the radioactivity is assumed to be on or near the surface, direct measurements are typically used to provide information on contaminant concentrations. Sometimes, however, it is necessary to collect actual samples of the building material surface for analysis in a laboratory.

7.5.2.1 Sample Volume

The sample volume collected from building surfaces is usually a less significant DQO concern than the area from which the sample was collected. This is because building surface DCGLs are usually expressed in terms of activity per unit area. It is still necessary to consider the sample volume to account for sample matrix effects that may reduce the chemical recovery, which in turn has an affect on the detection limit.

7.5.2.2 Sample Content

If residual activity is covered by paint or some other treatment, the underlying surface and the coating itself may be contaminated. If the activity is a pure alpha or low-energy beta emitter, measurements at the surface will probably not be representative of the actual residual activity level. In this case the surface layer is removed from the known area, such as by using a commercial stripping agent or by physically abrading the surface. The removed coating material is analyzed for activity content and the level converted to appropriate units (*i.e.*, Bq/m^2, $dpm/100\ cm^2$) for comparison with surface activity DCGLs. Direct measurements can be performed on the underlying surface after removal of the coating.

Residual radioactivity may be incorporated into building materials, such as pieces of concrete or other unusual matrices. Development of SOPs for collecting these types of samples may involve consultation with the analytical laboratory to help ensure that the objectives of the survey are achieved.

The thickness of the layer of building surface to be removed as a sample should be consistent with the development of the conceptual site model and the DCGLs. For most sites the surface layer will only be the first few millimeters of the material being sampled.

7.5.2.3 Sampling Equipment

Tools used to provide samples of building surfaces depend on the material to be sampled. Concrete may require chisels, hammers, drills, or other tools specifically designed to remove a thin layer of the surface. Wood surfaces may require using a sander or a saw to collect a sample. Paint may be chemically or physically stripped from the surface.

Sample containers for these samples are generally the same as those recommended for soil samples. If chemicals are used to strip paint or other surface materials, the chemical resistance of the container should be considered.

7.5.3 Other Media

Surface soil and building surfaces are the media addressed in MARSSIM during the final status survey design. Other media may be involved and may have been remediated. Data collection activities during preliminary surveys (*i.e.*, scoping, characterization, remedial action support) may involve collecting samples of other media to support the final status survey design. Examples of other media that may be sampled include:

- subsurface soil
- ground water
- surface water
- sediments
- sewers and septic systems
- flora and fauna (plants and animals)
- airborne particulates
- air (gas)

Appendix M provides a list of resources that can be used to develop sample collection procedures for other media that may required by preliminary surveys to support the development of a final status survey design.

7.6 Field Sample Preparation and Preservation

Proper sample preparation and preservation are essential parts of any radioactivity sampling program. The sampling objectives should be specified before sampling activities begin. Precise records of sample collection and handling are necessary to ensure that data obtained from different locations or time frames are correctly compared.

The appropriateness of sample preparation techniques is a function of the analysis to be performed (EPA 1992a, 1992b). Field sample preparation procedures are a function of the specified analysis and the objectives of the survey. It is essential that these objectives be clearly established and agreed upon in the early stages of survey planning (see Section 2.3).

7.6.1 Surface Soil

Soil and sediment samples, in most protocols, require no field preparation and are not preserved. In some protocols, cooling of soil samples to 4 °C is required during shipping and storage of soil samples. This is not a practice normally followed for the radiochemical analysis of soil samples.

When replicate samples are prepared in the field, it is necessary to homogenize the sample prior to separation into replicates. There are standard procedures for homogenizing soil in the laboratory (ASTM 1995), but the equipment required for these procedures may not be available in the field. Simple field techniques, such as cone and quarter, or using a riffle splitter to divide the sample may be appropriate if the sample can be dried (ASTM 1993, EPA 1991g). If the sample contains significant amounts of residual water (*e.g.*, forms clumps of soil) and there are no facilities for drying the sample, it is recommended that the homogenization and separation into replicates be performed in a laboratory. It is preferable to use non-blind replicates where the same laboratory prepares and analyzes the replicates rather than use poorly homogenized or heterogeneous samples to prepare replicates samples.

7.6.2 Building Surfaces

Field preparation and preservation of building and associated materials, including smear samples, is not generally required. Homogenization of samples to prepare replicates is the same for building surface material and soil.

7.6.3 Other Media

Other media may have significant requirements related to field sample preparation and preservation. For example, water samples may need filtering and acidification. Storage at reduced temperatures (*i.e.*, cooling or freezing) to reduce biological activity may be necessary for some samples. Addition of chemical preservatives for specific radionuclides or media may also be required.

7.7 Analytical Procedures

The selection of the appropriate radioanalytical methods is normally made prior to the procurement of analytical services and is included in the statement-of-work of the request for proposal. The statement-of-work may dictate the use of specific methods or be performance based. Unless there is a regulatory requirement, such as conformance to the EPA drinking water methods (EPA 1980a), the specification of performance based methodology is encouraged. One reason for this is that a laboratory will usually perform better using the methods routinely employed in its laboratory as contrasted to using other methods with which it has less experience.

The laboratory is also likely to have historical data on performance for methods routinely used by that laboratory. However, the methods employed in a laboratory should be derived from a reliable source, such as those listed in Table 7.2.

Table 7.2 Examples of References for Routine Analytical Methods

- *Methods of Air Sampling and Analysis* (Lodge 1988)
- *Annual Book of ASTM Standards, Water and Environmental technology. Volume 11.04, Environmental Assessment; Hazardous Substances and Oil Spill Responses; Waste Management; Environmental Risk Assessment* (ASTM 1997)
- *Standard Methods for the Examination of Water and Wastewater* (APHA 1995)
- *EML Procedures Manual* (DOE 1990b)
- *Radiochemical Analytical Procedures for Analysis of Environmental Samples* (EPA 1979)
- *Radiochemistry Procedures Manual* (EPA 1984a)
- *Indoor Radon and Radon Decay Product Measurement Protocols* (EPA 1992d)
- *USAEHA Environmental Sampling Guide* (Department of the Army 1993)

This section briefly describes specific equipment and procedures to be used once the sample is prepared for analysis. The results of these analyses (*i.e.*, the levels of radioactivity found in these samples) are the values used to determine the level of residual activity at a site. In a decommissioning effort, the DCGLs are expressed in terms of the concentrations of certain radionuclides. It is of vital importance, therefore, that the analyses be accurate and of adequate sensitivity for the radionuclides of concern. The selection of analytical procedures should be coordinated with the laboratory and specified in the survey plan.

Analytical methods should be adequate to meet the data needs identified in the DQO process. Consultation with the laboratory performing the analysis is recommended before selecting a course of action. MARSSIM is not intended to limit the selection of analytical procedures, rather all applicable methods should be reviewed to provide results that meet the objectives of the survey. The decision maker and survey planning team should decide whether routine methods will be used at the site or if non-routine methods may be acceptable.

- Routine analytical methods are documented with information on minimum performance characteristics, such as detection limit, precision and accuracy, and useful range of radionuclide concentrations and sample sizes. Routine methods may be issued by a recognized organization (*e.g.*, Federal or State agency, professional organization), published in a refereed journal, or developed by an individual laboratory. Table 7.2 lists examples of sources for routine methods.

- Non-routine methods address situations with unusual or problematic matrices, low detection limits, or new parameters, procedures or techniques. Non-routine methods include adjustments to routine methods, new techniques published in refereed literature, and development of new methods.

References that provide information on radiochemical methodology and should be considered in the methods review and selection process are available from such organizations as:

- National Council on Radiation Protection and Measurements (NCRP)
- American Society of Testing and Materials (ASTM)
- Radiological and Environmental Sciences Laboratory (RESL), Idaho Falls, Idaho (Operated by the DOE)
- DOE Technical Measurements Center, Grand Junction, CO
- Environmental Measurements Laboratory (EML); formerly the Health and Safety Laboratory of the DOE

Equipment vendor literature, catalogs, and instrument manuals are often a source of useful information on the characteristics of radiation detection equipment. Table 7.3 provides a summary of common laboratory methods with estimated detection limits.

Analytical procedures in the laboratory consist of several parts that are assembled to produce an SOP for a specific project or sample type. These parts include:

- laboratory sample preparation
- sample dissolution
- sample purification
- preparation for counting
- counting
- data reduction

Table 7.3 Typical Measurement Sensitivities for Laboratory Radiometric Procedures

Sample Type	Radionuclides or Radiation Measured	Procedure	Approximate Measurement Sensitivity
Smears (filter paper)	Gross alpha	Gas-flow proportional counter; 5-min count	5 dpm
		Alpha scintillation detector with scaler; 5-min count	20 dpm
	Gross beta	Gas-flow proportional counter; 5-min count	10 dpm
		End window GM with scaler; 5-min count (unshielded detector)	80 dpm
	Low energy beta (^{3}H, ^{14}C, ^{63}Ni)	Liquid scintillation spectrometer; 5-min count	30 dpm
Soil Sediment	^{137}Cs, ^{60}Co, ^{226}Ra (^{214}Bi)[a], ^{232}Th (^{228}Ac), ^{235}U	Germanium detector (25% relative efficiency) with multichannel analyzer; pulse height analyzer; 500-g sample; 15-min analysis	0.04-0.1 Bq/g (1-3 pCi/g)
	$^{234, 235, 238}U$; $^{238, 239, 240}Pu$; $^{227, 228, 230, 232}Th$; other alpha emitters	Alpha spectroscopy with multichannel analyzer - pyrosulfate fusion and solvent extraction; surface barrier detector; pulse height analyzer; 1-g sample; 16-hr count	0.004-0.02 Bq/g (0.1-0.5 pCi/g)
Water	Gross alpha	Gas-flow proportional counter; 100-ml sample, 200-min count	0.04 Bq/L (1 pCi/l)
	Gross beta	Gas-flow proportional counter; 100-ml sample, 200-min count	0.04 Bq/L (1 pCi/L)
	^{137}Cs, ^{60}Co, ^{226}Ra (^{214}Bi), ^{232}Th (^{228}Ac), ^{235}U	Germanium detector (25% relative efficiency) with multichannel analyzer; pulse height analyzer; 3.5L sample, 16-hr count	0.4 Bq/L (10 pCi/L)
	$^{234, 235, 238}U$; $^{238, 239, 240}Pu$; $^{227, 228, 230, 232}Th$; other alpha emitters	Alpha spectroscopy with multichannel analyzer - solvent extraction; surface barrier detector; pulse height analyzer; 100 ml sample, 30 min count	0.004-0.02 Bq/L (0.1-0.5 pCi/L)
	^{3}H	Liquid scintillation spectrometry; 5-ml sample, 30-min count	10 Bq/L (300 pCi/L)

[a] Indicates that a member of the decay series is measured to determine activity level of the parent radionuclide of primary interest.

7.7.1 Photon Emitting Radionuclides

There is no special sample preparation required for counting samples using a germanium detector or a sodium iodide detector beyond placing the sample in a known geometry for which the detector has been calibrated. The samples can be measured as they arrive at the laboratory, or the sample can be dried, ground to a uniform particle size, and mixed to provide a more homogeneous sample if required by the SOPs.

The samples are typically counted using a germanium detector with a multichannel analyzer or a sodium iodide detector with a multichannel analyzer. Germanium detectors have better resolution and can identify peaks (and the associated radionuclides) at lower concentrations. Sodium iodide detectors often have a higher efficiency and are significantly less expensive than germanium detectors. Low-energy photons (*i.e.*, x-rays and gamma rays below 50 keV) can be measured using specially designed detectors with an entrance window made from a very light metal, typically beryllium. Descriptions of germanium and sodium iodide detectors are provided in Appendix H.

Data reduction is usually the critical step in measuring photon emitting radionuclides. There are often several hundred individual gamma ray energies detected within a single sample. Computer software is usually used to identify the peaks, associate them with the proper energy, associate the energy with one or more radionuclides, correct for the efficiency of the detector and the geometry of the sample, and provide results in terms of concentrations with the associated uncertainty. It is important that the software be either a well-documented commercial package or thoroughly evaluated and documented before use.

7.7.2 Beta Emitting Radionuclides

Laboratory sample preparation is an important step in the analysis of surface soil and other solid samples for beta emitting radionuclides. The laboratory will typically have a sample preparation procedure that involves drying the sample and grinding the soil so that all of the particles are less than a specified size to provide a homogeneous sample. A small portion of the homogenized sample is usually all that is required for the individual analysis.

Once the sample has been prepared, a small portion is dissolved, fused, or leached to provide a clear solution containing the radionuclide of interest. The only way to ensure that the sample is solubilized is to completely dissolve the sample. However, this can be an expensive and time-consuming step in the analysis. In some cases, leaching with strong acids can consistently provide greater than 80% recovery of the radionuclide of interest (NCRP 1976a) and may be acceptable for certain applications. Gross beta measurements may be performed on material that has not been dissolved.

After dissolution, the sample is purified using a variety of chemical reactions to remove bulk chemical and radionuclide impurities. The objective is to provide a chemically and radiologically pure sample for measurement. Examples of purification techniques include precipitation, liquid-liquid extraction, ion-exchange chromatography, distillation, and electrodeposition. Gross beta measurements may be performed on material that has not been purified.

After the sample is purified, it is prepared for counting. Beta emitting radionuclides are usually prepared for a specific type of counter in a specified geometry. Solid material is usually precipitated and collected on a filter in a circular geometry to provide a homogeneous sample. Liquid samples are typically converted to the appropriate chemical form and diluted to a specified volume in preparation for counting.

Measurements of solid samples are typically performed using a gas-flow proportional counter. Because total beta activity is measured, it is important that the purification step be performed to remove any interfering radionuclides. Liquid samples are usually diluted using a liquid scintillation cocktail and counted using a liquid scintillation spectrometer. Liquid scintillation spectrometers can be used for low-energy beta emitting radionuclides, such as ^{3}H and ^{63}Ni. They also have high counting efficiencies, but often have a high instrument background as well. Gas-flow proportional counters have a very low background. Appendix H provides a description of both the gas-flow proportional counter and the liquid scintillation spectrometer.

Data reduction for beta emitting radionuclides is less complicated than that for photon emitting radionuclides. Since the beta detectors report total beta activity, the calculation to determine the concentration for the radionuclide of interest is straightforward.

7.7.3 Alpha Emitting Radionuclides

Laboratory sample preparation for alpha emitting radionuclides is similar to that for beta emitting radionuclides. Sample dissolution and purification tasks are also similar to those performed for beta emitting radionuclides.

Because of the limited penetrating power of alpha particles, the preparation for counting is often a critical step. Gross alpha measurements can be made using small sample sizes with a gas-flow proportional counter, but self-absorption of the alpha particles results in a relatively high detection limit for this technique. Liquid scintillation spectrometers can also be used to measure alpha emitting radionuclides but the resolution limits the usefulness of this technique. Most alpha emitting radionuclides are measured in a vacuum (to limit absorption by air) using alpha spectroscopy. This method requires that the sample be prepared as a virtually weightless mount in a specific geometry. Electrodeposition is the traditional method for preparing samples for counting. This technique provides the highest resolution, but it requires a significant amount of

training and expertise on the part of the analyst to produce a high quality sample. Precipitation of the radionuclide of interest on the surface of a substrate is often used to prepare samples for alpha spectroscopy. While this technique generally produces a spectrum with lower resolution, the preparation time is relatively short compared to electrodeposition, and personnel can be trained to prepare acceptable samples relatively quickly.

Alpha emitting radionuclides are typically measured using alpha spectroscopy. The data reduction requirements for alpha spectroscopy are greater than those for beta emitting radionuclides, and similar to those for photon emitting radionuclides. Alpha spectroscopy produces a spectrum of alpha particles detected at different energies, but because the sample is purified prior to counting, all of the alpha particles come from radionuclides of a single element. This simplifies the process of associating each peak with a specific radionuclide, but the lower resolution associated with alpha spectroscopy increases the difficulty of identifying the peaks. Although commercial software packages are available for interpreting alpha spectroscopy results, an experienced operator is required to ensure that the software is working properly.

7.8 Sample Tracking

Sample tracking refers to the identification of samples, their location, and the individuals responsible for their custody and transfer of the custody. This process covers the entire process from collection of the samples and remains intact through the analysis and final holding or disposal. It begins with the taking of a sample where its identification and designation of the sample are critical to being able to relate the analytical result to a site location.

Tracking samples from collection to receipt at the analytical laboratory is normally done through a Chain of Custody process, and documented on a Chain-of-Custody (COC) record. Once samples are received by the laboratory, internal tracking (*e.g.*, COC) procedures should be in place and codified through SOPs that assure integrity of the samples. Documentation of changes in the custody of a sample(s) is important. This is especially true for samples that may be used as evidence to establish compliance with a release criterion. In such cases, there should be sufficient evidence to demonstrate that the integrity of the sample is not compromised from the time it is collected to the time it is analyzed. During this time, the sample should either be under the positive control of a responsible individual or secured and protected from any activity that could change the true value of the results or the nature of the sample. When this degree of sample handling or custody is necessary, written procedures should be developed for field operations and for interfacing between the field operations and the analytical laboratory. This ensures that a clear transfer of the custodial responsibility is well documented and no questions exist as to who is responsible for the sample at any time.

7.8.1 Field Tracking Considerations

- Field personnel are responsible for maintaining field logbooks with adequate information to relate the sample identifier (sample number) to its location and for recording other information necessary to adequately interpret results of sample analytical data.
- The sample collector is responsible for the care and custody of the samples until they are properly transferred or dispatched. This means that samples are in their possession, under constant observation, or secured. Samples may be secured in a sealed container, locked vehicle, locked room, *etc.*
- Sample labels should be completed for each sample using waterproof ink.
- The survey manager or designee determines whether or not proper custody procedures were followed during the field work, and decides if additional sampling is indicated.
- If photographs are included as part of the sampling documentation, the name of the photographer, date, time, site location, and site description should be entered sequentially in a logbook as the photos are taken. After the photographs are developed, the prints should be serially numbered.

7.8.2 Transfer of Custody

- All samples leaving the site should be accompanied by a Chain-of-Custody record. This record documents sample custody transfer from the sampler, often through another person, to the laboratory. The individuals relinquishing the samples should sign and date the record. The record should include a list, including sample designation (number), of the samples in the shipping container and the analysis requested for each sample.
- Shipping containers should be sealed and include a tamper indicating seal that will indicate if the container seal has been disturbed. The method of shipment, courier name, or other pertinent information should be listed in the Chain-of-Custody record.
- The original Chain-of-Custody record should accompany the samples. A copy of the record should be retained by the individual or organization relinquishing the samples.
- Discuss the custody objectives with the shipper to ensure that the objectives are met. For example, if the samples are sent by mail and the originator of the sample requires a record that the shipment was delivered, the package should be registered with return receipt requested. If, on the other hand, the objective is to simply provide a written record of the shipment, a certificate of mailing may be a less expensive and appropriate alternative.
- The individual receiving the samples should sign and date the record. The condition of the container and the tamper indicating seal should be noted on the Chain-of-Custody record. Any problems with the individual samples, such as a broken container, should be noted on the record.

7.8.3 Laboratory Tracking

When the samples are received by the laboratory they are prepared for radiochemical analyses. This includes the fractionation of the sample into aliquots. The tracking and Chain-of-Custody documentation within the laboratory become somewhat complicated due to the fact that several portions of the original sample may exist in the laboratory at a given time. The use of a computer based Laboratory Information System (LIMS) can greatly assist in tracking samples and fractions through the analytical system.

The minimal laboratory tracking process consists of the following:

- transfer of custody on receipt of the samples (original Chain-of-Custody form is retained by the laboratory and submitted with the data package for the samples)
- documentation of sample storage (location and amount)
- documentation of removal and return of sample aliquots (amount, date and time, person removing or returning, and reason for removal)
- transfer of the samples and residues to the receiving authority (usually the site from which they were taken)

The procedure for accomplishing the above varies from laboratory to laboratory, but the exact details of performing the operations of sample tracking should be contained in a SOP.

7.9 Packaging and Transporting Samples

All samples being shipped for radiochemical analysis should be properly packaged and labeled before transport offsite or within the site. The primary concern is the possibility of spills, leaks, or breakage of the sample containers. In addition to resulting in the loss of samples and cross-contamination, the possible release of hazardous material poses a threat to the safety of persons handling and transporting the package.

Suggestions on packaging and shipping radioactive environmental samples are listed below.

1) Review NRC requirements (10 CFR part 71) and Department of Transportation (DOT) requirements (49 CFR parts 170 through 189) for packaging and shipping radioactive environmental samples.

2) Visually inspect each sample container for indication of leaks or defects in the sample container.

a) Liquid samples should be shipped in plastic containers, if possible, and the caps on the containers should be secured with tape. One exception to the use of plastic bottles is samples collected for ^{3}H analyses which may require glass containers.

b) Heavy plastic bags, with sealable tops, can be used to contain solid samples (*e.g.*, soil, sediment, air filters). The zip-lock should be secured with tape. Heavy plastic lawn bags can be used to contain vegetation samples. The tops should be closed with a "tie" that is covered by tape to prevent it from loosening and slipping off.

3) Wipe individual sample containers with a damp cloth or paper towel to remove any exterior contamination. The outer surfaces of containers holding samples collected in a contaminated area should be surveyed with a hand-held instrument(s), appropriate for the suspected type of radioactivity (β/γ or α).

4) If glass sample containers are used, place sample containers inside individual plastic bags and seal in order to contain the sample in case of breakage.

5) Use packing material (*e.g.*, paper, styrofoam, "bubble wrap") to immobilize and isolate each sample container and buffer hard knocks on the outer container during shipping. This is especially important in cold weather when plastic containers may become brittle and water samples may freeze.

6) When liquid samples are shipped, include a sufficient quantity of an absorbent material (*e.g.*, vermiculite) to absorb all liquid packed in the shipping container in case of breakage. This absorbent material may suffice as the packing material described above in item 5.

7) Include the original, signed and dated, Chain-of-Custody (COC) form, identifying each sample in the package. It is good practice to place the COC form in a plastic bag to prevent it from becoming wet or contaminated in case of a spill during shipment. If possible, avoid having multiple packages of samples covered by a single COC form.

8) Seal closed the package and apply COC tape in such a manner that it must be torn (broken) in order to open the package. The tape should carry the signature of the sender, and the date and time, so that it cannot be removed and replaced undetected.

9) Ice chests, constructed of metal or hard plastic, make excellent shipping containers for radioactive environmental samples.

If samples are sent offsite for analysis, the shipper is responsible for complying with all applicable Federal, State, and local regulations. Applicable Federal regulations are briefly addressed below. Any State or local regulation will very likely reflect a Federal regulation.

7.9.1 U.S. Nuclear Regulatory Commission Regulations

NRC regulations for packaging, preparation, and shipment of licensed material are contained in 10 CFR Part 71: "Packaging and Transportation of Radioactive materials".

Samples containing low levels of radioactivity are exempted as set forth in §§ 71.10. A licensee is exempt from all requirements of Part 71 if the specific activity of the sample being shipped is not greater than 74,000 Bq/kg (2,000 pCi/g).

Low Specific Activity Material (LSAM) is defined in §§ 71.4: "Definitions." Samples classified as LSAM need only meet the requirements of the U.S. Department of Transportation (DOT), discussed below, and the requirements of §§ 71.88: "Air transport of plutonium." Most environmental samples will fall into this category.

7.9.2 U.S. Department of Transportation Regulations

The U.S. Department of Transportation provides regulations governing the transport of hazardous materials under the Hazardous Materials Transportation Act of 1975 (88 Stat. 2156, Public Law 93-633). Applicable requirements of the regulations are found in 49 CFR Parts 170 through 189. Shippers of samples containing radioactivity should be aware of the current rules in the following areas.

- Accident Reporting - 49 CFR 171
- Marking and Labeling Packages for Shipment - 49 CFR 172
- Packaging - 49 CFR 173
- Placarding a Package - 49 CFR 172
- Registration of Shipper/Carrier - 49 CFR 107
- Shipper Required Training - 49 CFR 172
- Shipping Papers & Emergency Information - 49 CFR 172
- Transport by Air - 49 CFR 175

- Transport by Rail - 49 CFR 174
- Transport by Vessel - 49 CFR 176
- Transport on Public Highway - 49 CFR 177

7.9.3 U.S. Postal Service Regulations

Any package containing radioactive materials is nonmailable if required to bear the U.S. Department of Transportation's Radioactive White-1 (49 CFR 172.436), Radioactive Yellow-II (49 CFR 172.438), or Radioactive Yellow-III (49 CFR 172.440) label, or if it contains quantities of radioactive material in excess of those authorized in Publication 6, Radioactive Material, of the U.S. Postal Service.

8 INTERPRETATION OF SURVEY RESULTS

8.1 Introduction

This chapter discusses the interpretation of survey results, primarily those of the final status survey. Interpreting a survey's results is most straightforward when measurement data are entirely higher or lower than the $DCGL_W$. In such cases, the decision that a survey unit meets or exceeds the release criterion requires little in terms of data analysis. However, formal statistical tests provide a valuable tool when a survey unit's measurements are neither clearly above nor entirely below the $DCGL_W$. Nevertheless, the survey design *always* makes use of the statistical tests in helping to assure that the number of sampling points and the measurement sensitivity are adequate, but not excessive, for the decision to be made.

Section 8.2 discusses the assessment of data quality. The remainder of this chapter deals with application of the statistical tests used in the decision-making process, and the evaluation of the test results. In addition, an example checklist is provided to assist the user in obtaining the necessary information for interpreting the results of a final status survey.

8.2 Data Quality Assessment

Data Quality Assessment (DQA) is a scientific and statistical evaluation that determines if the data are of the right type, quality, and quantity to support their intended use. An overview of the DQA process appears in Section 2.3 and Appendix E. There are five steps in the DQA process:

- Review the Data Quality Objectives (DQOs) and Survey Design
- Conduct a Preliminary Data Review
- Select the Statistical Test
- Verify the Assumptions of the Statistical Test
- Draw Conclusions from the Data

The effort expended during the DQA evaluation should be consistent with the graded approach used in developing the survey design. More information on DQA is located in Appendix E, and the EPA Guidance Document QA/G-9 (EPA 1996a). Data should be verified and validated as described in Section 9.3 prior to the DQA evaluation.

8.2.1 Review the Data Quality Objectives (DQOs) and Sampling Design

The first step in the DQA evaluation is a review of the DQO outputs to ensure that they are still applicable. For example, if the data suggest the survey unit was misclassified as Class 3 instead of Class 1, then the original DQOs should be redeveloped for the correct classification.

The sampling design and data collection documentation should be reviewed for consistency with the DQOs. For example, the review should check that the appropriate number of samples were taken in the correct locations and that they were analyzed with measurement systems with appropriate sensitivity. Example checklists for different types of surveys are given in Chapter 5.

Determining that the sampling design provides adequate power is important to decision making, particularly in cases where the levels of residual radioactivity are near the $DCGL_W$. This can be done both prospectively, during survey design to test the efficacy of a proposed design, and retrospectively, during interpretation of survey results to determine that the objectives of the design are met. The procedure for generating power curves for specific tests is discussed in Appendix I. Note that the accuracy of a prospective power curve depends on estimates of the data variability, σ, and the number of measurements. After the data are analyzed, a sample estimate of the data variability, namely the sample standard deviation (s) and the actual number of valid measurements will be known. The consequence of inadequate power is that a survey unit that actually meets the release criterion has a higher probability of being incorrectly deemed *not* to meet the release criterion.

8.2.2 Conduct a Preliminary Data Review

To learn about the structure of the data—identifying patterns, relationships, or potential anomalies—one can review quality assurance (QA) and quality control (QC) reports, prepare graphs of the data, and calculate basic statistical quantities.

8.2.2.1 Data Evaluation and Conversion

Radiological survey data are usually obtained in units, such as the number of counts per unit time, that have no intrinsic meaning relative to DCGLs. For comparison of survey data to DCGLs, the survey data from field and laboratory measurements are converted to DCGL units. Further information on instrument calibration and data conversion is given in Section 6.2.7.

Basic statistical quantities that should be calculated for the sample data set are the:

- mean
- standard deviation
- median

Example:

Suppose the following 20 concentration values are from a survey unit:

90.7, 83.5, 86.4, 88.5, 84.4, 74.2, 84.1, 87.6, 78.2, 77.6,
86.4, 76.3, 86.5, 77.4, 90.3, 90.1, 79.1, 92.4, 75.5, 80.5.

First, the average of the data (83.5) and the sample standard deviation (5.7) should be calculated.

The average of the data can be compared to the reference area average and the $DCGL_W$ to get a preliminary indication of the survey unit status. Where remediation is inadequate, this comparison may readily reveal that a survey unit contains excess residual radioactivity—even before applying statistical tests. For example, if the average of the data exceeds the $DCGL_W$ and the radionuclide of interest does not appear in background, then the survey unit clearly does not meet the release criterion. On the other hand, if every measurement in the survey unit is below the $DCGL_W$, the survey unit clearly meets the release criterion.[1]

The value of the sample standard deviation is especially important. If too large compared to that assumed during the survey design, this may indicate an insufficient number of samples were collected to achieve the desired power of the statistical test. Again, inadequate power can lead to unnecessary remediation.

The median is the middle value of the data set when the number of data points is odd, and is the average of the two middle values when the number of data points is even. Thus 50% of the data points are above the median, and 50% are below the median. Large differences between the mean and the median would be an early indication of skewness in the data. This would also be evident in a histogram of the data. For the example data above, the median is 84.25 (*i.e.*, (84.1 + 84.4)/2). The difference between the median and the mean (*i.e.*, 84.25 - 83.5 = 0.75) is a small fraction of the sample standard deviation (*i.e.*, 5.7). Thus, in this instance, the mean and median would not be considered significantly different.

Examining the minimum, maximum, and range of the data may provide additional useful information. The minimum in this example is 74.2 and the maximum is 92.4, so the range is 92.4 - 74.2 = 18.2. This is only 3.2 standard deviations. Thus, the range is not unusually large. When there are 30 or fewer data points, values of the range much larger than about 4 to 5 standard deviations would be unusual. For larger data sets the range might be wider.

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[1] It can be verified that if every measurement is below the $DCGL_W$, the conclusion from the statistical tests will always be that the survey unit does not exceed the release criterion.

8.2.2.2 Graphical Data Review

At a minimum, a graphical data review should consist of a posting plot and a histogram. Quantile plots are also useful diagnostic tools, particularly in the two-sample case, to compare the survey unit and reference area. These are discussed in Appendix I, Section I.8.

A *posting plot* is simply a map of the survey unit with the data values entered at the measurement locations. This potentially reveals heterogeneities in the data—especially possible patches of elevated residual radioactivity. Even in a reference area, a posting plot can reveal spatial trends in background data that might affect the results of the two-sample statistical tests.

If the data above were obtained using a triangular grid in a rectangular survey unit, the posting plot might resemble the display in Figure 8.1. Figure 8.1a shows no unusual patterns in the data. Figure 8.1b shows a different plot of the same values, but with individual results associated with different locations within the survey unit. In this plot there is an obvious trend towards smaller values as one moves from left to right across the survey unit. This trend is not apparent in the simple initial listing of the data. The trend may become more apparent if isopleths are added to the posting plot.

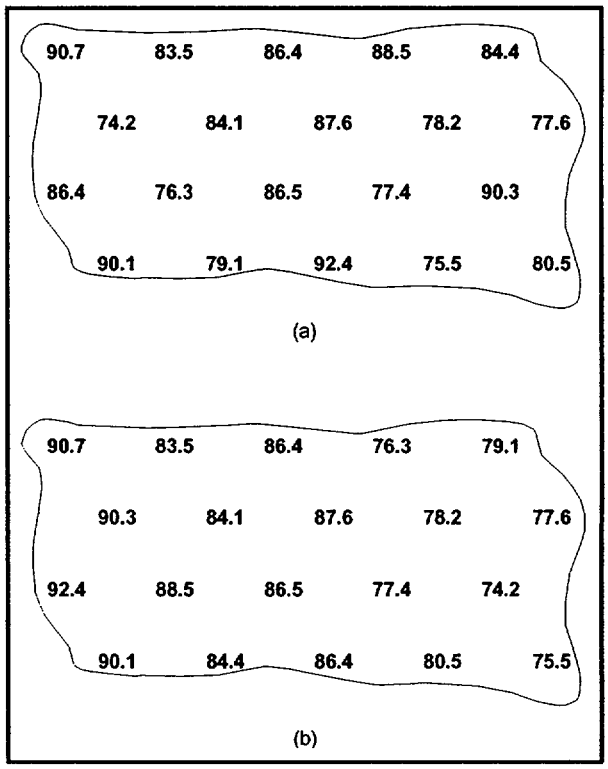

Figure 8.1 Examples of Posting Plots

If the posting plot reveals systematic spatial trends in the survey unit, the cause of the trends would need to be investigated. In some cases, such trends could be due to residual radioactivity, but may also be due to inhomogeneities in the survey unit background. Other diagnostic tools for examining spatial data trends may be found in EPA Guidance Document QA/G-9 (EPA 1996a). The use of geostatistical tools to evaluate spatial data trends may also be useful in some cases (EPA 1989a).

A *frequency plot* (or a histogram) is a useful tool for examining the general shape of a data distribution. This plot is a bar chart of the number of data points within a certain range of values. A frequency plot of the example data is shown in Figure 8.2). A simple method for generating a

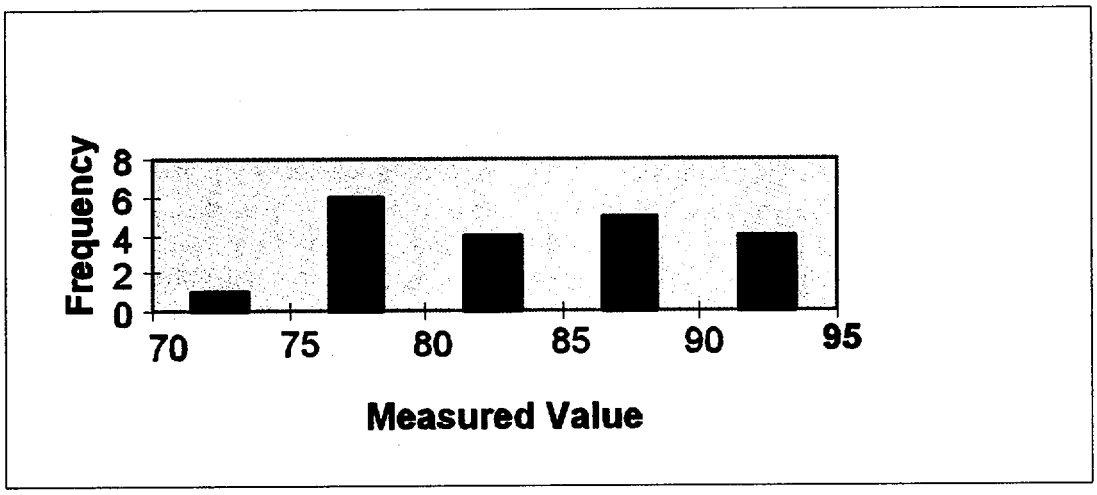

Figure 8.2 Example of a Frequency Plot

rough frequency plot is the stem and leaf display discussed in Appendix I, Section I.7. The frequency plot will reveal any obvious departures from symmetry, such as skewness or bimodality (two peaks), in the data distributions for the survey unit or reference area. The presence of two peaks in the survey unit frequency plot may indicate the existence of isolated areas of residual radioactivity. In some cases it may be possible to determine an appropriate background for the survey unit using this information. The interpretation of the data for this purpose will generally be highly dependent on site-specific considerations and should only be pursued after a consultation with the responsible regulatory agency.

The presence of two peaks in the background reference area or survey unit frequency plot may indicate a mixture of background concentration distributions due to different soil types, construction materials, *etc.* The greater variability in the data due to the presence of such a mixture will reduce the power of the statistical tests to detect an adequately remediated survey unit. These situations should be avoided whenever possible by carefully matching the background reference areas to the survey units, and choosing survey units with homogeneous backgrounds.

Skewness or other asymmetry can impact the accuracy of the statistical tests. A data transformation (*e.g.*, taking the logarithms of the data) can sometimes be used to make the distribution more symmetric. The statistical tests would then be performed on the transformed data. When the underlying data distribution is highly skewed, it is often because there are a few high areas. Since the EMC is used to detect such measurements, the difference between using the median and the mean as a measure for the degree to which uniform residual radioactivity remains in a survey unit tends to diminish in importance.

8.2.3 Select the Tests

An overview of the statistical considerations important for final status surveys appears in Section 2.5 and Appendix D. The most appropriate procedure for summarizing and analyzing the data is chosen based on the preliminary data review. The parameter of interest is the mean concentration in the survey unit. The nonparametric tests recommended in this manual, in their most general form, are tests of the median. If one assumes that the data are from a symmetric distribution—where the median and the mean are effectively equal—these are also tests of the mean. If the assumption of symmetry is violated, then nonparametric tests of the median approximately test the mean. Computer simulations (*e.g.*, Hardin and Gilbert, 1993) have shown that the approximation is a good one. That is, the correct decision will be made about whether or not the mean concentration exceeds the DCGL, even when the data come from a skewed distribution. In this regard, the nonparametric tests are found to be correct more often than the commonly used Student's t test. The robust performance of the Sign and WRS tests over a wide range of conditions is the reason that they are recommended in this manual.

When a given set of assumptions is true, a parametric test designed for exactly that set of conditions will have the highest power. For example, if the data are from a normal distribution, the Student's t test will have higher power than the nonparametric tests. It should be noted that for large enough sample sizes (*e.g.*, large number of measurements), the Student's t test is not a great deal more powerful than the nonparametric tests. On the other hand, when the assumption of normality is violated, the nonparametric tests can be very much more powerful than the t test. Therefore, any statistical test may be used provided that the data are consistent with the assumptions underlying their use. When these assumptions are violated, the prudent approach is to use the nonparametric tests which generally involve fewer assumptions than their parametric equivalents.

The one-sample statistical test (Sign test) described in Section 5.5.2.3 should only be used if the contaminant is not present in background and radionuclide-specific measurements are made. The one-sample test may also be used if the contaminant is present at such a small fraction of the $DCGL_W$ value as to be considered insignificant. In this case, background concentrations of the radionuclide are included with the residual radioactivity (*i.e.*, the entire amount is attributed to facility operations). Thus, the total concentration of the radionuclide is compared to the release criterion. This option should only be used if one expects that ignoring the background concentration will not affect the outcome of the statistical tests. The advantage of ignoring a small background contribution is that no reference area is needed. This can simplify the final status survey considerably.

The one-sample Sign test (Section 8.3.1) evaluates whether the median of the data is above or below the $DCGL_W$. If the data distribution is symmetric, the median is equal to the mean. In cases where the data are severely skewed, the mean may be above the $DCGL_W$, while the median

is below the $DCGL_W$. In such cases, the survey unit does *not* meet the release criterion regardless of the result of the statistical tests. On the other hand, if the largest measurement is below the $DCGL_W$, the Sign test will always show that the survey unit meets the release criterion.

For final status surveys, the two-sample statistical test (Wilcoxon Rank Sum test, discussed in Section 5.5.2.2) should be used when the radionuclide of concern appears in background or if measurements are used that are not radionuclide specific. The two-sample Wilcoxon Rank Sum (WRS) test (Section 8.4.1) assumes the reference area and survey unit data distributions are similar except for a possible shift in the medians. When the data are severely skewed, the value for the mean difference may be above the $DCGL_W$, while the median difference is below the $DCGL_W$. In such cases, the survey unit does *not* meet the release criterion regardless of the result of the statistical test. On the other hand, if the difference between the largest survey unit measurement and the smallest reference area measurement is less than the $DCGL_W$, the WRS test will always show that the survey unit meets the release criterion.

8.2.4 Verify the Assumptions of the Tests

An evaluation to determine that the data are consistent with the underlying assumptions made for the statistical procedures helps to validate the use of a test. One may also determine that certain departures from these assumptions are acceptable when given the actual data and other information about the study. The nonparametric tests described in this chapter assume that the data from the reference area or survey unit consist of independent samples from each distribution.

Spatial dependencies that potentially affect the assumptions can be assessed using posting plots (Section 8.2.2.2). More sophisticated tools for determining the extent of spatial dependencies are also available (*e.g.*, EPA QA/G-9). These methods tend to be complex and are best used with guidance from a professional statistician.

Asymmetry in the data can be diagnosed with a stem and leaf display, a histogram, or a Quantile plot. As discussed in the previous section, data transformations can sometimes be used to minimize the effects of asymmetry.

One of the primary advantages of the nonparametric tests used in this report is that they involve fewer assumptions about the data than their parametric counterparts. If parametric tests are used, (*e.g.*, Student's *t* test), then any additional assumptions made in using them should be verified (*e.g.*, testing for normality). These issues are discussed in detail in EPA QA/G-9 (EPA 1996a).

One of the more important assumptions made in the survey design described in Chapter 5 is that the sample sizes determined for the tests are sufficient to achieve the data quality objectives set for the Type I (α) and Type II (β) error rates. Verification of the power of the tests ($1-\beta$) to detect adequate remediation may be of particular interest. Methods for assessing the power are discussed in Appendix I.9. If the hypothesis that the survey unit residual radioactivity exceeds the release criterion is accepted, there should be reasonable assurance that the test is equally effective in determining that a survey unit has residual contamination less than the $DCGL_W$. Otherwise, unnecessary remediation may result. For this reason, it is better to plan the surveys cautiously—even to the point of:

- overestimating the potential data variability
- taking too many samples
- overestimating minimum detectable concentrations (MDCs)

If one is unable to show that the DQOs were met with reasonable assurance, a resurvey may be needed. Examples of assumptions and possible methods for their assessment are summarized in Table 8.1.

Table 8.1 Methods for Checking the Assumptions of Statistical Tests

Assumption	Diagnostic
Spatial Independence	Posting Plot
Symmetry	Histogram, Quantile Plot
Data Variance	Sample Standard Deviation
Power is Adequate	Retrospective Power Chart

8.2.5 Draw Conclusions from the Data

The types of measurements that can be made in a survey unit are 1) direct measurements at discrete locations, 2) samples collected at discrete locations, and 3) scans. The statistical tests are only applied to measurements made at discrete locations. Specific details for conducting the statistical tests are given in Sections 8.3 and 8.4. When the data clearly show that a survey unit meets or exceeds the release criterion, the result is often obvious without performing the formal statistical analysis. Table 8.2 describes examples of circumstances leading to specific conclusions based on a simple examination of the data.

Table 8.2 Summary of Statistical Tests

Radionuclide not in background and radionuclide-specific measurements made:

Survey Result	Conclusion
All measurements less than $DCGL_W$	Survey unit meets release criterion
Average greater than $DCGL_W$	Survey unit does not meet release criterion
Any measurement greater than $DCGL_W$ and the average less than $DCGL_W$	Conduct Sign test and elevated measurement comparison

Radionuclide in background or radionuclide non-specific (gross) measurements made:

Survey Result	Conclusion
Difference between largest survey unit measurement and smallest reference area measurement is less than $DCGL_W$	Survey unit meets release criterion
Difference of survey unit average and reference area average is greater than $DCGL_W$	Survey unit does not meet release criterion
Difference between any survey unit measurement and any reference area measurement greater than $DCGL_W$ and the difference of survey unit average and reference area average is less than $DCGL_W$	Conduct WRS test and elevated measurement comparison

Both the measurements at discrete locations and the scans are subject to the elevated measurement comparison (EMC). The result of the EMC is not conclusive as to whether the survey unit meets or exceeds the release criterion, but is a flag or trigger for further investigation. The investigation may involve taking further measurements to determine that the area and level of the elevated residual radioactivity are such that the resulting dose or risk meets the release criterion.[2] The investigation should also provide adequate assurance, using the DQO process, that there are no other undiscovered areas of elevated residual radioactivity in the survey unit that might otherwise result in a dose or risk exceeding the release criterion. In some cases, this may lead to re-classifying all or part of a survey unit—unless the results of the investigation indicate that reclassification is not necessary. The investigation level appropriate for each class of survey unit and type of measurement is shown in Table 5.8 and is described in Section 5.5.2.6.

[2] Rather than, or in addition to, taking further measurements the investigation may involve assessing the adequacy of the exposure pathway model used to obtain the DCGLs and area factors, and the consistency of the results obtained with the Historical Site Assessment and the scoping, characterization and remedial action support surveys.

8.2.6 Example

To illustrate the data interpretation process, consider an example facility with 14 survey units consisting of interior concrete surfaces, one interior survey unit with drywall surfaces, and two exterior survey units. The contaminant of concern is ^{60}Co. The interior surfaces were measured with a gas-flow proportional counter (see Appendix H) with an active surface area of 20 cm^2 to determine total beta-gamma activity. Because these measurements are not radionuclide specific, appropriate reference areas were chosen for comparison. The exterior soil was measured with a germanium spectrometer to provide radionuclide-specific results. A reference area is not needed because ^{60}Co does not have a significant background in soil.

The exterior Class 3 survey unit incorporates areas that are not expected to contain residual radioactivity. The exterior Class 2 survey unit is similar to the Class 3 survey unit, but is expected to contain residual radioactivity below the $DCGL_W$. The Class 1 Interior Concrete survey units are expected to contain small areas of elevated activity that may or may not exceed the $DCGL_W$. The Class 2 Interior Drywall survey unit is similar to the Class 1 Interior Concrete survey unit, but the drywall is expected to have a lower background, less measurement variability, and a more uniform distribution of contamination. The Class 2 survey unit is not expected to contain areas of activity above the $DCGL_W$. Section 8.3 describes the Sign test used to evaluate the survey units where the contaminant is not present in background. Section 8.4 describes the WRS test used to evaluate the survey units where the contaminant is present in background. Section 8.5 discusses the evaluation of the results of the statistical tests and the decision regarding compliance with the release criterion. The survey design parameters and DQOs developed for these survey units are summarized in Table 8.3.

Table 8.3 Final Status Survey Parameters for Example Survey Units

Survey Unit	Type	DQO		$DCGL_W$	Estimated Standard Deviation, σ		Test/Section
		α	β		Survey	Reference	
Interior Concrete	Class 1	.05	.05	5000 dpm per 100 cm^2	625 dpm per 100 cm^2	220 dpm per 100 cm^2	WRS/App. A
Interior Drywall	Class 2	.025	.05	5000 dpm per 100 cm^2	200 dpm per 100 cm^2	200 dpm per 100 cm^2	WRS/8.4.3
Exterior Lawn	Class 2	.025	.025	140 Bq/kg	3.8 Bq/kg	N/A	Sign/8.3.3
Exterior Lawn	Class 3	.025	.01	140 Bq/kg	3.8 Bq/kg	N/A	Sign/8.3.4

8.3 Contaminant Not Present in Background

The statistical test discussed in this section is used to compare each survey unit directly with the applicable release criterion. A reference area is not included because the measurement technique is radionuclide-specific and the radionuclide of concern is not present in background (see Section 8.2.6). In this case the contaminant levels are compared directly with the $DCGL_W$. The method in this section should only be used if the contaminant is not present in background or is present at such a small fraction of the $DCGL_W$ value as to be considered insignificant. In addition, one-sample tests are applicable only if radionuclide-specific measurements are made to determine the concentrations. Otherwise, the method in Section 8.4 is recommended.

Reference areas and reference samples are not needed when there is sufficient information to indicate there is essentially no background concentration for the radionuclide being considered. With only a single set of survey unit samples, the statistical test used here is called a one-sample test. See Section 5.5 for further information appropriate to following the example and discussion presented here.

8.3.1 One-Sample Statistical Test

The Sign test is designed to detect uniform failure of remedial action throughout the survey unit. This test does not assume that the data follow any particular distribution, such as normal or log-normal. In addition to the Sign Test, the $DCGL_{EMC}$ (see Section 5.5.2.4) is compared to each measurement to ensure none exceeds the $DCGL_{EMC}$. If a measurement exceeds this DCGL, then additional investigation is recommended, at least locally, to determine the actual areal extent of the elevated concentration.

The hypothesis tested by the Sign test is

Null Hypothesis
H_0: The median concentration of residual radioactivity in the survey unit is greater than the $DCGL_W$

versus

Alternative Hypothesis
H_a: The median concentration of residual radioactivity in the survey unit is less than the $DCGL_W$

The null hypothesis is assumed to be true unless the statistical test indicates that it should be rejected in favor of the alternative. The null hypothesis states that the probability of a measurement less than the $DCGL_W$ is less than one-half, *i.e.*, the 50th percentile (or median) is

greater than the $DCGL_W$. Note that some individual survey unit measurements may exceed the $DCGL_W$ even when the survey unit as a whole meets the release criterion. In fact, a survey unit average that is close to the $DCGL_W$ might have almost half of its individual measurements greater than the $DCGL_W$. Such a survey unit may still not exceed the release criterion.

The assumption is that the survey unit measurements are independent random samples from a symmetric distribution. If the distribution of measurements is symmetric, the median and the mean are the same.

The hypothesis specifies a release criterion in terms of a $DCGL_W$. The test should have sufficient power (1-β, as specified in the DQOs) to detect residual radioactivity concentrations at the Lower Boundary of the Gray Region (LBGR). If σ is the standard deviation of the measurements in the survey unit, then Δ/σ expresses the size of the shift (*i.e.*, $\Delta = DCGL_W - LBGR$) as the number of standard deviations that would be considered "large" for the distribution of measurements in the survey unit. The procedure for determining Δ/σ is given in Section 5.5.2.3.

8.3.2 Applying the Sign Test

The Sign test is applied as outlined in the following five steps, and further illustrated by the examples in Sections 8.3.3 and 8.3.4.

1. List the survey unit measurements, X_i, $i = 1, 2, 3..., N$.

2. Subtract each measurement, X_i, from the $DCGL_W$ to obtain the differences:
   $D_i = DCGL_W - X_i$, $i = 1, 2, 3..., N$.

3. Discard each difference that is exactly zero and reduce the sample size, N, by the number of such zero measurements.

4. Count the number of positive differences. The result is the test statistic S+. Note that a positive difference corresponds to a measurement below the $DCGL_W$ and contributes evidence that the survey unit meets the release criterion.

5. Large values of S+ indicate that the null hypothesis (that the survey unit exceeds the release criterion) is false. The value of S+ is compared to the critical values in Table I.3. If S+ is greater than the critical value, k, in that table, the null hypothesis is rejected.

8.3.3 Sign Test Example: Class 2 Exterior Soil Survey Unit

For the Class 2 Exterior Soil survey unit, the one-sample nonparametric statistical test is appropriate since the radionuclide of concern does not appear in background and radionuclide-specific measurements were made.

Table 8.3 shows that the DQOs for this survey unit include $\alpha = 0.025$ and $\beta = 0.025$. The $DCGL_W$ is 140 Bq/kg (3.8 pCi/g) and the estimated standard deviation of the measurements is σ = 3.8 Bq/kg (0.10 pCi/g). Since the estimated standard deviation is much smaller than the $DCGL_W$, the LBGR should be set so that Δ/σ is about 3.

If $\Delta/\sigma = (DCGL_W - LBGR)/\sigma$
$= 3$
then $LBGR = DCGL_W - 3\sigma$
$= 140 - (3 \times 3.8)$
$= 128$ Bq/kg (3.5 pCi/g).

Table 5.5 indicates the number of measurements estimated for the Sign Test, N, is 20 ($\alpha = 0.025$, $\beta = 0.025$, and $\Delta/\sigma = 3$). (Table I.2a in Appendix I also lists the number of measurements estimated for the Sign test.) This survey unit is Class 2, so the 20 measurements needed were made on a random-start triangular grid. When laying out the grid, 22 measurement locations were identified.

The 22 measurements taken on the exterior lawn Class 2 survey unit are shown in the first column of Table 8.4. The mean of these data is 129 Bq/kg (3.5 pCi/g) and the standard deviation is 11 Bq/kg (0.30 pCi/g). Since the number of measurements is even, the median of the data is the average of the two middle values (126+128)/2 = 127 Bq/kg (3.4 pCi/g). A Quantile Plot of the data is shown in Appendix I.8, Figure I.3.

There are five measurements that exceed the $DCGL_W$ value of 140 Bq/kg: 142, 143, 145, 148, and 148. However, none exceed the mean of the data plus three standard deviations: $127 + (3 \times 11) = 160$ Bq/kg (4.3 pCi/g). Thus, these values appear to reflect the overall variability of the concentration measurements rather than to indicate an area of elevated activity—*provided* that these measurements were scattered through the survey unit. However, if a posting plot demonstrates that the locations of these measurements are grouped together, then that portion of the survey unit containing these locations merits further investigation.

The middle column of Table 8.4 contains the differences, $DCGL_W$ - *Data*, and the last column contains the signs of the differences. The bottom row shows the number of measurements with positive differences, which is the test statistic S+. In this case, S+ = 17.

The value of S+ is compared to the appropriate critical value in Table I.3. In this case, for $N = 22$ and $\alpha = 0.025$, the critical value is 16. Since S+ = 17 exceeds this value, the null hypothesis that the survey unit exceeds the release criterion is rejected.

Table 8.4 Example Sign Analysis: Class 2 Exterior Soil Survey Unit

Data (Bq/kg)	$DCGL_W$-Data (Bq/kg)	Sign
121	19	1
143	-3	-1
145	-5	-1
112	28	1
125	15	1
132	8	1
122	18	1
114	26	1
123	17	1
148	-8	-1
115	25	1
113	27	1
126	14	1
134	6	1
148	-8	-1
130	10	1
119	21	1
136	4	1
128	12	1
125	15	1
142	-2	-1
129	11	1
Number of positive differences S+ = 17		

8.3.4 Sign Test Example: Class 3 Exterior Soil Survey Unit

For the Class 3 exterior soil survey unit, the one-sample nonparametric statistical test is again appropriate since the radionuclide of concern does not appear in background and radionuclide-specific measurements were made.

Table 8.3 shows that the DQOs for this survey unit include $\alpha = 0.025$ and $\beta = 0.01$. The $DCGL_W$ is 140 Bq/kg (3.8 pCi/g) and the estimated standard deviation of the measurements is $\sigma = 3.8$ Bq/kg (0.10 pCi/g). Since the estimated standard deviation is much smaller than the $DCGL_W$, the lower bound for the gray region should be set so that Δ/σ is about 3.

If $\Delta/\sigma = (DCGL_W - LBGR)/\sigma$
$= 3$
then $LBGR = DCGL_W - 3\sigma$
$= 140 - (3 \times 4)$
$= 128$ Bq/kg (3.5 pCi/g).

Table 5.5 indicates that the sample size estimated for the Sign Test, N, is 23 ($\alpha = 0.025$, $\beta = 0.01$, and $\Delta/\sigma = 3$). This survey unit is Class 3, so the measurements were made at random locations within the survey unit.

The 23 measurements taken on the exterior lawn are shown in the first column of Table 8.5. Notice that some of these measurements are negative (-0.37 in cell A6). This might occur if an analysis background (*e.g.*, the Compton continuum under a spectrum peak) is subtracted to obtain the net concentration value. The data analysis is both easier and more accurate when numerical values are reported *as obtained* rather than reporting the results as "less than" or not detected. The mean of these data is 2.1 Bq/kg (0.057 pCi/g) and the standard deviation is 3.3 Bq/kg (0.089 pCi/g). None of the data exceed $2.1 + (3 \times 3.3) = 12.0$ Bq/kg (0.32 pCi/g). Since N is odd, the median is the middle (12^{th} highest) value, namely 2.6 Bq/kg (0.070 pCi/g).

An initial review of the data reveals that every data point is below the $DCGL_W$, so the survey unit meets the release criterion specified in Table 8.3. For purely illustrative purposes, the Sign test analysis is performed. The middle column of Table 8.5 contains the quantity $DCGL_W$ - Data. Since every data point is below the $DCGL_W$, the sign of $DCGL_W$ - Data is always positive. The number of positive differences is equal to the number of measurements, N, and so the Sign test statistic S+ is 23. The null hypothesis will always be rejected at the maximum value of S+ (which in this case is 23) and the survey unit passes. Thus, the application of the Sign test in such cases requires no calculations and one need not consult a table for a critical value. If the survey is properly designed, the critical value must always be less than N.

Passing a survey unit without making a single calculation may seem an unconventional approach. However, the key is in the survey design which is intended to ensure enough measurements are made to satisfy the DQOs. As in the previous example, after the data are collected the conclusions and power of the test can be checked by constructing a retrospective power curve as outlined in Appendix I, Section I..9.

One final consideration remains regarding the survey unit classification: "Was any definite amount of residual radioactivity found in the survey unit?" This will depend on the MDC of the measurement method. Generally the MDC is at least 3 or 4 times the estimated measurement standard deviation. In the present case, the largest observation, 9.3 Bq/kg (0.25 pCi/g), is less than three times the estimated measurement standard deviation of 3.8 Bq/kg (0.10 pCi/g). Thus, it is unlikely that any of the measurements could be considered indicative of positive contamination. This means that the Class 3 survey unit classification was appropriate.

Table 8.5 Sign Test Example Data for Class 3 Exterior Survey Unit

	A	B	C
1	Data	$DCGL_W$-Data	Sign
2	3.0	137.0	1
3	3.0	137.0	1
4	1.9	138.1	1
5	0.37	139.6	1
6	-0.37	140.4	1
7	6.3	133.7	1
8	-3.7	143.7	1
9	2.6	137.4	1
10	3.0	137.0	1
11	-4.1	144.1	1
12	3.0	137.0	1
13	3.7	136.3	1
14	2.6	137.4	1
15	4.4	135.6	1
16	-3.3	143.3	1
17	2.1	137.9	1
18	6.3	133.7	1
19	4.4	135.6	1
20	-0.37	140.4	1
21	4.1	135.9	1
22	-1.1	141.1	1
23	1.1	138.9	1
24	9.3	130.7	1
25	Number of positive differences S+ =		23

If one determines that residual radioactivity is definitely present, this would indicate that the survey unit was initially mis-classified. Ordinarily, MARSSIM recommends a resurvey using a Class 1 or Class 2 design. If one determines that the survey unit is a Class 2, a resurvey might be avoided if the survey unit does not exceed the maximum size for such a classification. In this case, the only difference in survey design would be whether the measurements were obtained on a random or on a triangular grid. Provided that the initial survey's scanning methodology is sufficiently sensitive to detect areas at $DCGL_W$ without the use of an area factor, this difference in the survey grids alone would not affect the outcome of the statistical analysis. Therefore, if the above conditions were met, a resurvey might not be necessary.

8.4 Contaminant Present in Background

The statistical tests discussed in this section will be used to compare each survey unit with an appropriately chosen, site-specific reference area. Each reference area should be selected on the basis of its similarity to the survey unit, as discussed in Section 4.5.

8.4.1 Two-Sample Statistical Test

The comparison of measurements from the reference area and survey unit is made using the Wilcoxon Rank Sum (WRS) test (also called the Mann-Whitney test). The WRS test should be conducted for each survey unit. In addition, the EMC is performed against each measurement to ensure that it does not exceed a specified investigation level. If any measurement in the remediated survey unit exceeds the specified investigation level, then additional investigation is recommended, at least locally, regardless of the outcome of the WRS test.

The WRS test is most effective when residual radioactivity is uniformly present throughout a survey unit. The test is designed to detect whether or not this activity exceeds the $DCGL_W$. The advantage of the nonparametric WRS test is that it does not assume that the data are normally or log-normally distributed. The WRS test also allows for "less than" measurements to be present in the reference area and the survey units. As a general rule, the WRS test can be used with up to 40 percent "less than" measurements in either the reference area or the survey unit. However, the use of "less than" values in data reporting is not recommended as discussed in Section 2.3.5. When possible, report the actual result of a measurement together with its uncertainty.

The hypothesis tested by the WRS test is

Null Hypothesis
H_0: The median concentration in the survey unit exceeds that in the reference area by more than the $DCGL_W$

versus

Alternative Hypothesis
H_a: The median concentration in the survey unit exceeds that in the reference area by less than the $DCGL_W$

The null hypothesis is assumed to be true unless the statistical test indicates that it should be rejected in favor of the alternative. One assumes that any difference between the reference area and survey unit concentration distributions is due to a shift in the survey unit concentrations to higher values (*i.e.*, due to the presence of residual radioactivity in addition to background). Note that some or all of the survey unit measurements may be larger than some reference area

measurements, while still meeting the release criterion. Indeed, some survey unit measurements may exceed some reference area measurements by more than the $DCGL_W$. The result of the hypothesis test determines whether or not the survey unit as a whole is deemed to meet the release criterion. The EMC is used to screen individual measurements.

Two assumptions underlying this test are: 1) samples from the reference area and survey unit are independent, identically distributed random samples, and 2) each measurement is independent of every other measurement, regardless of the set of samples from which it came.

8.4.2 Applying the Wilcoxon Rank Sum Test

The WRS test is applied as outlined in the following six steps and further illustrated by the examples in Section 8.4.3 and Appendix A.

1. Obtain the adjusted reference area measurements, Z_i, by adding the $DCGL_W$ to each reference area measurement, X_i. $Z_i = X_i + DCGL_W$

2. The m adjusted reference sample measurements, Z_i, from the reference area and the n sample measurements, Y_i, from the survey unit are pooled and ranked in order of increasing size from 1 to N, where $N = m+n$.

3. If several measurements are tied (*i.e.*, have the same value), they are all assigned the average rank of that group of tied measurements.

4. If there are t "less than" values, they are all given the average of the ranks from 1 to t. Therefore, they are all assigned the rank $t(t+1)/(2t) = (t+1)/2$, which is the average of the first t integers. If there is more than one detection limit, all observations below the largest detection limit should be treated as "less than" values.[3]

5. Sum the ranks of the adjusted measurements from the reference area, W_r. Note that since the sum of the first N integers is $N(N+1)/2$, one can equivalently sum the ranks of the measurements from the survey unit, W_s, and compute $W_r = N(N+1)/2 - W_s$.

6. Compare W_r with the critical value given in Table I.4 for the appropriate values of n, m, and α. If W_r is greater than the tabulated value, reject the hypothesis that the survey unit exceeds the release criterion.

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[3] If more than 40 percent of the data from either the reference area or survey unit are "less than," the WRS test *cannot* be used. Such a large proportion of non-detects suggest that the DQO process be re-visited for this survey to determine if the survey unit was properly classified or the appropriate measurement method was used. As stated previously, the use of "less than" values in data reporting is not recommended. Wherever possible, the actual result of a measurement, together with its uncertainty, should be reported.

8.4.3 Wilcoxon Rank Sum Test Example: Class 2 Interior Drywall Survey Unit

In this example, the gas-flow proportional counter measures total beta-gamma activity (see Appendix H) and the measurements are not radionuclide specific. The two-sample nonparametric test is appropriate for the Class 2 interior drywall survey unit because gross beta-gamma activity contributes to background even though the radionuclide of interest does not appear in background.

Table 8.3 shows that the DQOs for this survey unit include $\alpha = 0.025$ and $\beta = 0.05$. The $DCGL_W$ is 8,300 Bq/m^2 (5,000 dpm per 100 cm^2) and the estimated standard deviation of the measurements is about $\sigma = 1040$ Bq/m^2 (625 dpm per 100 cm^2). The estimated standard deviation is 8 times less than the $DCGL_W$. With this level of precision, the width of the gray region can be made fairly narrow. As noted earlier, sample sizes do not decrease very much once Δ/σ exceeds 3 or 4. In this example, the lower bound for the gray region was set so that Δ/σ is about 4.

If $\Delta/\sigma = (DCGL_W - LBGR)/\sigma$
$= 4$

then $LBGR = DCGL_W - 4\sigma$
$= 3{,}000 - (4 \times 375)$
$= 4{,}200$ Bq/m^2 (2,500 dpm per 100 cm^2).

In Table 5.3, one finds that the number of measurements estimated for the WRS test is 11 in each survey unit and 11 in each reference area ($\alpha = 0.025$, $\beta = 0.05$, and $\Delta/\sigma = 4$). (Table I.2b in Appendix I also lists the number of measurements estimated for the WRS test.) This survey unit was classified as Class 2, so the 11 measurements needed in the survey unit and the 11 measurements needed in the reference area were made using a random-start triangular grid.[4]

Table 8.6 lists the data obtained from the gas-flow proportional counter in units of counts per minute. A reading of 160 cpm with this instrument corresponds to the $DCGL_W$ of 8,300 Bq/m^2 (5,000 dpm per 100 cm^2). Column A lists the measurement results as they were obtained. The average and standard deviation of the reference area measurements are 44 and 4.4 cpm, respectively. The average and standard deviation of the survey unit measurements are 98 and 5.3 cpm, respectively.

[4]A random start systematic grid is used in Class 2 and 3 survey units primarily to limit the size of any potential elevated areas. Since areas of elevated activity are not an issue in the reference areas, the measurement locations can be either random or on a random start systematic grid (see Section 5.5.2.5).

Table 8.6 WRS Test for Class 2 Interior Drywall Survey Unit

	A	B	C	D	E
1	Data (cpm)	Area	Adjusted Data	Ranks	Reference Area Ranks
2	49	R	209	22	22
3	35	R	195	12	12
4	45	R	205	17.5	17.5
5	45	R	205	17.5	17.5
6	41	R	201	14	14
7	44	R	204	16	16
8	48	R	208	21	21
9	37	R	197	13	13
10	46	R	206	19	19
11	42	R	202	15	15
12	47	R	207	20	20
13	104	S	104	9.5	0
14	94	S	94	4	0
15	98	S	98	6	0
16	99	S	99	7	0
17	90	S	90	1	0
18	104	S	104	9.5	0
19	95	S	95	5	0
20	105	S	105	11	0
21	93	S	93	3	0
22	101	S	101	8	0
23	92	S	92	2	0
24	Sum =			253	**187**

In column B, the code "R" denotes a reference area measurement, and "S" denotes a survey unit measurement. Column C contains the Adjusted Data. The Adjusted Data are obtained by adding the $DCGL_W$ to the reference area measurements (see Section 8.4.2, Step 1). The ranks of the adjusted data appear in Column D. They range from 1 to 22, since there is a total of 11+11 measurements (see Section 8.4.2, Step 2).

Note that there were two cases of measurements tied with the same value, at 104 and 209. Each tied measurement is always assigned the average of the ranks. Therefore, both measurements at 104, are assigned rank (9+10)/2 = 9.5 (see Section 8.4.2, Step 3). Also note that the sum of *all* of the ranks is still 22(22+1)/2 = 253. Checking this value with the formula in Step 5 of Section 8.4.2 is recommended to guard against errors in the rankings.

Column E contains only the ranks belonging to the reference area measurements. The total is 187. This is compared with the entry for the critical value of 156 in Table I.4 for $\alpha = 0.025$, with $n = 11$ and $m = 11$. Since the sum of the reference area ranks is greater than the critical value, the null hypothesis (*i.e.*, that the average survey unit concentration exceeds the $DCGL_W$) is rejected.

The analysis for the WRS test is very well suited to the use of a computer spreadsheet. The spreadsheet formulas used for the example above are given in Appendix I.10, Table I.11.

8.4.4 Class 1 Interior Concrete Survey Unit

As in the previous example, the gas-flow proportional counter measures total beta-gamma activity (see Appendix H) and the measurements are not radionuclide specific. The two-sample nonparametric test is appropriate for the Class 1 interior concrete survey unit because gross beta-gamma activity contributes to background even though the radionuclide of interest does not appear in background.

Appendix A provides a detailed description of the calculations for the Class 1 interior concrete survey unit.

8.4.5 Multiple Radionuclides

The use of the unity rule when there is more than one radionuclide to be considered is discussed in Appendix I.11. An example application appears in Section I.11.4.

8.5 Evaluating the Results: The Decision

Once the data and the results of the tests are obtained, the specific steps required to achieve site release depend on the procedures instituted by the governing regulatory agency and site-specific ALARA considerations. The following suggested considerations are for the interpretation of the test results with respect to the release limit established for the site or survey unit. Note that the tests need not be performed in any particular order.

8.5.1 Elevated Measurement Comparison

The Elevated Measurement Comparison (EMC) consists of comparing each measurement from the survey unit with the *investigation levels* discussed in Section 5.5.2.6 and Section 8.2.5. The EMC is performed for both measurements obtained on the systematic-sampling grid and for locations flagged by scanning measurements. Any measurement from the survey unit that is equal to or greater than an *investigation level* indicates an area of relatively high concentrations that should be investigated—regardless of the outcome of the nonparametric statistical tests.

The statistical tests may not reject H_0 when only a very few high measurements are obtained in the survey unit. The use of the EMC against the investigation levels may be viewed as assurance that unusually large measurements will receive proper attention regardless of the outcome of those tests and that any area having the potential for significant dose contributions will be identified. The EMC is intended to flag potential failures in the remediation process. This should not be considered the primary means to identify whether or not a site meets the release criterion.

The derived concentration guideline level for the EMC is:

$$DCGL_{EMC} = A_m \times DCGL_W \qquad 8\text{-}1$$

where A_m is the area factor for the area of the systematic grid area. Note that $DCGL_{EMC}$ is an *a priori* limit, established both by the $DCGL_W$ and by the survey design (*i.e.*, grid spacing and scanning MDC). The true extent of an area of elevated activity can only be determined after performing the survey and taking additional measurements. Upon the completion of further investigation, the *a posteriori* limit, $DCGL_{EMC} = A_m \times DCGL_W$, can be established using the value of A_m appropriate for the *actual area of elevated concentration*. The area of elevated activity is generally bordered by concentration measurements below the $DCGL_W$. An individual elevated measurement on a systematic grid could conceivably represent an area four times as large as the systematic grid area used to define the $DCGL_{EMC}$. This is the area bounded by the nearest neighbors of the elevated measurement location. The results of the investigation should show that the appropriate $DCGL_{EMC}$ is not exceeded. Area factors are discussed in Section 5.5.2.4.

If measurements above the stated scanning MDC are found by sampling or by direct measurement at locations that were not flagged by the scanning survey, this may indicate that the scanning method did not meet the DQOs.

The preceding discussion primarily concerns Class 1 survey units. Measurements exceeding $DCGL_W$ in Class 2 or Class 3 areas may indicate survey unit mis-classification. Scanning coverage for Class 2 and Class 3 survey units is less stringent than for Class 1. If the investigation levels of Section 8.2.5 are exceeded, an investigation should: 1) ensure that the area of elevated activity discovered meets the release criterion, and 2) provide reasonable assurance that other undiscovered areas of elevated activity do not exist. If further investigation determines that the survey unit was mis-classified with regard to contamination potential, then a resurvey using the method appropriate for the new survey unit classification may be appropriate.

8.5.2 Interpretation of Statistical Test Results

The result of the statistical test is the decision to reject or not to reject the null hypothesis. Provided that the results of investigations triggered by the EMC were resolved, a rejection of the null hypothesis leads to the decision that the survey unit meets the release criterion. However, estimating the average residual radioactivity in the survey unit may also be necessary so that dose or risk calculations can be made. This estimate is designated δ (see Appendix D, Section D.6). The average concentration is generally the best estimator for δ (EPA 1992g).

If residual radioactivity is found in an isolated area of elevated activity—in addition to residual radioactivity distributed relatively uniformly across the survey unit—the unity rule (Section 4.3.3) can be used to ensure that the total dose is within the release criterion:

$$\frac{\ddot{a}}{DCGL_W} + \frac{(average\ concentration\ in\ elevated\ area - \ddot{a})}{(area\ factor\ for\ elevated\ area)(DCGL_W)} < 1 \qquad 8\text{-}2$$

If there is more than one elevated area, a separate term should be included for each. As an alternative to the unity rule, the dose or risk due to the actual residual radioactivity distribution can be calculated if there is an appropriate exposure pathway model available. Note that these considerations generally apply only to Class 1 survey units, since areas of elevated activity should not exist in Class 2 or Class 3 survey units.

A retrospective power analysis for the test will often be useful, especially when the null hypothesis is not rejected (see Appendix I.9). When the null hypothesis is not rejected, it may be because it is in fact true, or it may be because the test did not have sufficient power to detect that it is not true. The power of the test will be primarily affected by changes in the actual number of measurements obtained and their standard deviation. An effective survey design will slightly overestimate both the number of measurements and the standard deviation to ensure adequate power. This insures that a survey unit is not subjected to additional remediation simply because the final status survey is not sensitive enough to detect that residual radioactivity is below the guideline level. When the null hypothesis is rejected, the power of the test becomes a somewhat moot question. Nonetheless, even in this case, a retrospective power curve can be a useful diagnostic tool and an aid to designing future surveys.

8.5.3 If the Survey Unit Fails

The guidance provided in MARSSIM is fairly explicit concerning the steps that should be taken to show that a survey unit meets release criteria. Less has been said about the procedures that should be used if at any point the survey unit fails. This is primarily because there are many different ways that a survey unit may fail the final status survey. The overall level of residual

radioactivity may not pass the nonparametric statistical tests. Further investigation following the elevated measurement comparison may show that there is a large enough area with a concentration too high to meet the release criterion. Investigation levels may have caused locations to be flagged during scanning that indicate unexpected levels of residual radioactivity for the survey unit classification. Site-specific information is needed to fully evaluate all of the possible reasons for failure, their causes, and their remedies.

When a survey unit fails to demonstrate compliance with the release criterion, the first step is to review and confirm the data that led to the decision. Once this is done, the DQO Process (Appendix D) can be used to identify and evaluate potential solutions to the problem. The level of residual radioactivity in the survey unit should be determined to help define the problem. Once the problem has been stated the decision concerning the survey unit should be developed into a decision rule. Next, determine the additional data, if any, needed to document that the survey unit demonstrates compliance with the release criterion. Alternatives to resolving the decision statement should be developed for each survey unit that fails the tests. These alternatives are evaluated against the DQOs, and a survey design that meets the objectives of the project is selected.

For example, a Class 2 survey unit passes the nonparametric statistical tests, but has several measurements on the sampling grid that exceed the $DCGL_W$. This is unexpected in a Class 2 area, and so these measurements are flagged for further investigation. Additional sampling confirms that there are several areas where the concentration exceeds the $DCGL_W$. This indicates that the survey unit was mis-classified. However, the scanning technique that was used was sufficient to detect residual radioactivity at the $DCGL_{EMC}$ calculated for the sample grid. No areas exceeding the $DCGL_{EMC}$ where found. Thus, the only difference between the final status survey actually done, and that which would be required for a Class 1 area, is that the scanning may not have covered 100% of the survey unit area. In this case, one might simply increase the scan coverage to 100%. Reasons why the survey unit was misclassified should be noted. If no areas exceeding the $DCGL_{EMC}$ are found, the survey unit essentially demonstrates compliance with the release criterion as a Class 1 survey unit.

If, in the example above, the scanning technique was not sufficiently sensitive, it may be possible to re-classify as Class 1 only that portion of the survey unit containing the higher measurements. This portion would be re-sampled at the higher measurement density required for a Class 1 survey unit, with the rest of the survey unit remaining Class 2.

A second example might be a Class 1 Survey unit that passes the nonparametric statistical tests and contains some areas that were flagged for investigation during scanning. Further investigation, sampling and analysis indicates one area is truly elevated. This area has a concentration that exceeds the $DCGL_W$ by a factor greater than the area factor calculated for its actual size. This area is then remediated. Remediation control sampling shows that the residual

radioactivity was removed, and no other areas were contaminated with removed material. In this case one may simply document the original final status survey, the fact that remediation was performed, the results of the remedial action support survey, and the additional remediation data. In some cases, additional final status survey data may not be needed to demonstrate compliance with the release criterion.

As a last example, consider a Class 1 area which fails the nonparametric statistical tests. Confirmatory data indicates that the average concentration in the survey unit does exceed the $DCGL_W$ over a majority of its area. This indicates remediation of the entire survey unit is necessary, followed by another final status survey. Reasons for performing a final status survey in a survey unit with significant levels of residual radioactivity should be noted.

These examples are meant to illustrate the actions that may be necessary to secure the release of a survey unit that has failed to meet the release criterion. The DQO Process should be revisited to plan how to attain the original objective, that is to safely release the survey unit by showing that it meets the release criterion. Whatever data are necessary to meet this objective will be in addition to the final status survey data already in hand.

8.5.4 Removable Activity

Some regulatory agencies may require that smear samples be taken at indoor grid locations as an indication of removable surface activity. The percentage of removable activity assumed in the exposure pathway models has a great impact on dose calculations. However, measurements of smears are very difficult to interpret quantitatively. Therefore, the results of smear samples should not be used for determining compliance. Rather, they should be used as a diagnostic tool to determine if further investigation is necessary.

8.6 Documentation

Documentation of the final status survey should provide a complete and unambiguous record of the radiological status of the survey unit relative to the established DCGLs. In addition, sufficient data and information should be provided to enable an independent evaluation of the results of the survey including repeating measurements at some future time. The documentation should comply with all applicable regulatory requirements. Additional information on documentation is provided in Chapter 3, Chapter 5, Chapter 9, and Appendix N.

Much of the information in the final status report will be available from other decommissioning documents. However, to the extent practicable, this report should be a stand-alone document with minimum information incorporated by reference. This document should describe the

instrumentation or analytical methods used, how the data were converted to DCGL units, the process of comparing the results to the DCGLs, and the process of determining that the data quality objectives were met.

The results of actions taken as a consequence of individual measurements or sample concentrations in excess of the investigation levels should be reported together with any additional data, remediation, or re-surveys performed to demonstrate that issues concerning potential areas of elevated activity were resolved. The results of the data evaluation using statistical methods to determine if release criteria were satisfied should be described. If criteria were not met or if results indicate a need for additional data, appropriate further actions should be determined by the site management in consultation with the responsible regulatory agency.

EXAMPLE DATA INTERPRETATION CHECKLIST

CONVERT DATA TO STANDARD UNITS

_____ Structure activity in Bq/m^2 ($dpm/100\ cm^2$)
_____ Solid media (soil, *etc.*) activity in Bq/kg (pCi/g)

EVALUATE ELEVATED MEASUREMENTS

_____ Identify elevated data
_____ Compare data with derived elevated area criteria
_____ Determine need to remediate and/or reinvestigate elevated condition
_____ Compare data with survey unit classification criteria
_____ Determine need to investigate and/or reclassify

ASSESS SURVEY DATA

_____ Review DQOs and survey design
_____ Verify that data of adequate quantity and quality were obtained
_____ Perform preliminary assessments (graphical methods) for unusual or suspicious trends or results—investigate further as appropriate

PERFORM STATISTICAL TESTS

_____ Select appropriate tests for category of contaminant
_____ Conduct tests
_____ Compare test results against hypotheses
_____ Confirm power level of tests

COMPARE RESULTS TO GUIDELINES

_____ Determine average or median concentrations
_____ Confirm that residual activity satisfies guidelines

COMPARE RESULTS WITH DQOs*

_____ Determine whether all DQOs are satisfied
_____ Explain/describe deviations from design-basis DQOs

* ALARA may be included in the DQOs.

9 QUALITY ASSURANCE AND QUALITY CONTROL

9.1 Introduction

The goal of quality assurance and quality control (QA/QC) is to identify and implement sampling and analytical methodologies which limit the introduction of error into analytical data. For MARSSIM data collection and evaluation, a system is needed to ensure that radiation surveys produce results that are of the type and quality needed and expected for their intended use. A *quality system* is a management system that describes the elements necessary to plan, implement, and assess the effectiveness of QA/QC activities. This system establishes many functions including: quality management policies and guidelines for the development of organization- and project-specific quality plans; criteria and guidelines for assessing data quality; assessments to ascertain effectiveness of QA/QC implementation; and training programs related to QA/QC implementation. A quality system ensures that MARSSIM decisions will be supported by sufficient data of adequate quality and usability for their intended purpose, and further ensures that such data are authentic, appropriately documented, and technically defensible.

Any organization collecting and evaluating data for a particular program must be concerned with the quality of results. The organization must have results that: meet a well-defined need, use, or purpose; comply with program requirements; and reflect consideration of cost and economics. To meet the objective, the organization should control the technical, administrative, and human factors affecting the quality of results. Control should be oriented toward the appraisal, reduction, elimination, and prevention of deficiencies that affect quality.

Quality systems already exist for many organizations involved in the use of radioactive materials. There are self-imposed internal quality management systems (*e.g.*, DOE) or there are systems required by regulation by another entity (*e.g.*, NRC) which require a quality system as a condition of the operating license.[1] These systems are typically called Quality Assurance Programs. An organization may also obtain services from another organization that already has a quality system in place. When developing an organization-specific quality system, there is no need to develop new quality management systems, to the extent that a facility's current Quality Assurance Program can be used. Standard ANSI/ASQC E4-1994 (ASQC 1995) provides national consensus quality standards for environmental programs. It addresses both quality systems and the collection and evaluation of environmental data. Annex B of ANSI/ASQC E4-1994

[1] Numerous quality assurance and quality control (QA/QC) requirements and guidance documents have been applied to environmental programs. Until now, each Federal agency has developed or chosen QA/QC requirements to fit its particular mission and needs. Some of these requirements include DOE Order 5700.6c (DOE 1991c); EPA QA/R-2 (EPA 1994f); EPA QA/R-5 (EPA 1994c); 10 CFR 50, App. B; NUREG-1293, Rev. 1 (NRC 1991); Reg Guide 4.15 (NRC 1979); and MIL-Q-9858A (DOD 1963). In addition, there are several consensus standards for QA/AC, including ASME NQA-1 (ASME 1989), and ISO 9000/ASQC Q9000 series (ISO 1987). ANSI/ASQC E4-1994 (ASQC 1995) is a consensus standard specifically for environmental data collection.

(ASQC 1995) and Appendix K of MARSSIM illustrate how existing quality system documents compare with organization- and project-specific environmental quality system documents.

Table 9.1 illustrates elements of a quality system as they relate to the Data Life Cycle. Applying a quality system to a project is typically done in three phases as described in Section 2.3: 1) the planning phase where the Data Quality Objectives (DQOs) are developed following the process described in Appendix D and documented in the Quality Assurance Project Plan (QAPP),[2] 2) the implementation phase involving the collection of environmental data in accordance with approved procedures and protocols, and 3) the assessment phase including the verification and validation of survey results as discussed in Section 9.3 and the evaluation of the environmental data using Data Quality Assessment (DQA) as discussed in Section 8.2 and Appendix E.
Detailed guidance on quality systems is not provided in MARSSIM because a quality system should be in place and functioning prior to beginning environmental data collection activities.

Table 9.1 The Elements of a Quality System Related to the Data Life Cycle

Data Life Cycle	Quality System Elements
Planning	Data Quality Objectives (DQOs) Quality Assurance Project Plans (QAPPs) Standard Operating Procedures (SOPs)
Implementation	QAPPs SOPs Data collection Assessments and audits
Assessment	Data validation and verification Data Quality Assessment (DQA)

A graded approach bases the level of controls on the intended use of the results and the degree of confidence needed in their quality. Applying a graded approach may mean that some organizations (*e.g.*, those using the simplified procedures in Appendix B) make use of existing plans and procedures to conduct surveys. For many other organizations, the need for cleanup and restoration of contaminated facilities may create the need for one or more QAPPs suitable to the special needs of environmental data gathering, especially as it relates to the demonstration of compliance with regulatory requirements. There may even be a need to update or revise an existing quality management system.

[2] The quality assurance project plan is sometimes abbreviated QAPjP. MARSSIM adopts the terminology and abbreviations used in ANSI/ASQC E4-1994 (ASQC 1995) and EPA QA/R-5 (EPA 1994c).

9.2 Development of a Quality Assurance Project Plan

The Quality Assurance Project Plan (QAPP)[3] is the critical planning document for any environmental data collection operation because it documents how QA/QC activities will be implemented during the life cycle of a project (EPA 1997a). The QAPP is the blueprint for identifying how the quality system of the organization performing the work is reflected in a particular project and in associated technical goals. This section provides information on how to develop a QAPP based on the DQO process. The results of the DQO process provide key inputs to the QAPP and will largely determine the level of detail in the QAPP.

The consensus standard ANSI/ASQC E4-1994 (ASQC 1995) describes the minimum set of quality elements required to conduct programs involving environmental data collection and evaluation. Table 9.2 lists the quality elements for collection and evaluation of environmental data from ANSI/ASQC E4-1994. These quality elements are provided as examples that should be addressed when developing a QAPP. This table also includes references for obtaining additional information on each of these quality elements. Many of these elements will be addressed in existing documents, such as the organization's Quality Assurance Program or Quality Management Plan. Each of these quality elements should be considered during survey planning to determine the degree to which they will be addressed in the QAPP. Additional quality elements may need to be added to this list as a result of organizational preferences or requirements of Federal and State regulatory authorities. For example, safety and health or public participation may be included as elements to be considered during the development of a QAPP.

The QAPP should be developed using a graded approach as discussed in Section 9.1. In other words, existing procedures and survey designs can be included by reference. This is especially useful for sites using a simplified survey design process (*e.g.*, surveys designed using Appendix B).

A QAPP should be developed to document the results of the planning phase of the Data Life Cycle (see Section 2.3). The level of detail provided in the QAPP for relevant quality elements is determined using the DQO process during survey planning activities. Information that is already provided in existing documents does not need to be repeated in the QAPP, and can be included by reference (EPA 1997a).

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[3] MARSSIM uses the term Quality Assurance Project Plan to describe a single document that incorporates all of the elements of the survey design. This term is consistent with ANSI/ASQC E4-1994 (ASQC 1995) and EPA guidance (EPA 1994c, EPA 1997a), and is recommended to promote consistency. The use of the term QAPP in MARSSIM does not exclude the use of other terms (*e.g.*, Decommissioning Plan, Sampling and Analysis Plan, Field Sampling Plan) to describe survey planning documentation as long as the information in the documentation supports the objectives of the survey.

Table 9.2 Examples of QAPP Elements for Site Surveys and Investigations

QAPP Element	Information Source	
Planning and Scoping (reference the QA Manual for information on the quality system)	ASQC 1995 EPA 1994c EPA 1997a NRC 1997c EPA 1993d	Part A, Sections 2.1 and 2.7; Part B, Section 3.1 Sections A4, A5, A6 and A7 Chapter III, Sections A4, A5, A6, and A7 Chapter 14 Project Objectives
Design of Data Collection Operations (including training)	ASQC 1995 EPA 1994c EPA 1997a EPA 1993d	Part A, Section 2.3; Part B, Section 3.2 Sections A9 and B1 Chapter III, Sections A9 and B1 Sampling Design
Implementation of Planned Operations (including documents and records)	ASQC 1995 EPA 1994c EPA 1997a NRC 1997c EPA 1993d	Part A, Section 2.8; Part B, Section 3.3 Sections A1, A2, A3, B2, B3, B4, B5, B6, B7, B8, B9, and B10 Chapter III, Sections A1, A2, A3, B2, B3, B4, B5, B6, B7, B8, B9, and B10 Chapter 5 Sampling Execution, Sample Analysis
Assessment and Response	ASQC 1995 EPA 1994c EPA 1997a EPA 1993d	Part A, Section 2.9, Part B, Section 3.4 Sections C1 and C2 Chapter III, Sections C1 and C2 Exhibit 3, Reference Box 3
Assessment and Verification of Data Usability	ASQC 1995 EPA 1994c EPA 1997a NRC 1997c EPA 1993d	Part B, Section 3.5 Sections D1, D2, and D3 Chapter III, Sections D1, D2, and D3 Chapter 20, Appendix J, Appendix Q Assessment of Data Quality

For example, the quality system description, personnel qualifications and requirements, and Standard Operating Procedures (SOPs) for the laboratory analysis of samples may simply be references to existing documents (*e.g.*, Quality Management Plan, Laboratory Procedure Manual). SOPs for performing direct measurements with a specific instrument may be attached to the QAPP because this information may not be readily available from other sources.

There is no particular format recommended for developing a QAPP. Figure 9.1 provides an example of a QAPP format presented in EPA QA/R-5 (EPA 1994c). Appendix K compares the quality elements presented in this example to the quality elements found in EPA QAMS-005-80 (EPA 1980d), ASME NQA-1 (ASME 1989), DOE Order 5700.6c (DOE 1991c), MIL-Q-9858A (DOD 1963), and ISO 9000 (ISO 1987).

Project Management
- Title and Approval Sheet
- Table of Contents
- Distribution List
- Project/Task Organization
- Problem Definition/Background
- Project Task Description
- Quality Objectives and Criteria for Measurement Data
- Special Training Requirements/Certification

Measurement/Data Acquisition
- Sampling Process Design (Experimental Design)
- Sampling Methods Requirements
- Sample Handling and Custody Requirements
- Analytical Methods Requirements
- Quality Control Requirements
- Instrument/Equipment Testing, Inspection, and Maintenance Requirements
- Instrument Calibration and Frequency
- Inspection/Acceptance Requirements for Supplies and Consumables

Assessment/Oversight
- Assessments and Response Actions
- Reports to Management

Data Validation and Usability
- Data Review, Validation, and Verification Requirements
- Validation and Verification Methods
- Reconciliation with User Requirements

Figure 9.1 Example of a QAPP Format

9.3 Data Assessment

Assessment of environmental data is used to evaluate whether the data meet the objectives of the survey, and whether the data are sufficient to determine compliance with the DCGL (EPA 1992a, 1992b, 1996a). The assessment phase of the Data Life Cycle consists of three phases: data verification, data validation, and Data Quality Assessment (DQA). This section provides guidance on verifying and validating data collected during a final status survey designed to demonstrate compliance with a dose- or risk-based regulation. Guidance on DQA is provided in Chapter 8 and Appendix E. As with all components of a successful survey, the level of effort associated with the assessment of survey data should be consistent with the objectives of the survey (*i.e.*, a graded approach).

9.3.1 Data Verification

Data verification ensures that the requirements stated in the planning documents (*e.g.*, Quality Assurance Project Plan, Standard Operating Procedures) are implemented as prescribed. This means that deficiencies or problems that occur during implementation should be documented and reported. This also means that activities performed during the implementation phase are assessed regularly with findings documented and reported to management. Corrective actions undertaken should be reviewed for adequacy and appropriateness and documented in response to the findings. Data verification activities should be planned and documented in the QAPP. These assessments may include but are not limited to inspections, QC checks, surveillance, technical reviews, performance evaluations, and audits.

To ensure that conditions requiring corrective actions are identified and addressed promptly, data verification activities should be initiated as part of data collection during the implementation phase of the survey. The performance of tasks by personnel is generally compared to a prescribed method documented in the SOPs, and is generally assessed using inspections, surveillance, or audits. Self-assessments and independent assessments may be planned, scheduled, and performed as part of the survey. Self-assessment also means that personnel doing work should document and report deficiencies or problems that they encounter to their supervisors or management.

The performance of equipment such as radiation detectors or measurement systems such as an instrument and human operator can be monitored using control charts. Control charts are used to record the results of quantitative QC checks such as background and daily calibration or performance checks. Control charts document instrument and measurement system performance on a regular basis and identify conditions requiring corrective actions on a real time basis. Control charts are especially useful for surveys that extend over a significant period of time (*e.g.*, weeks instead of days) and for equipment that is owned by a company that is frequently used to collect survey data. Surveys that are accomplished in one or two days and use rented instruments may not benefit significantly from the preparation and use of control charts. The use of control charts is usually documented in the SOPs.

A technical review is an independent assessment that provides an in-depth analysis and evaluation of documents, activities, material, data, or items that require technical verification to ensure that established requirements are satisfied (ASQC 1995). A technical review typically requires a significant effort in time and resources and may not be necessary for all surveys. A complex survey using a combination of scanning, direct measurements, and sampling for multiple survey units is more likely to benefit from a detailed technical review than a simple survey design calling for relatively few measurements using one or two measurement techniques for a single survey unit.

9.3.2 Data Validation

Data validation activities ensure that the results of data collection activities support the objectives of the survey as documented in the QAPP, or support a determination that these objectives should be modified. Data Usability is the process of ensuring or determining whether the quality of the data produced meets the intended use of the data (EPA 1992a, EPA 1997a). Data verification compares the collected data with the prescribed activities documented in the SOPs; data validation compares the collected data to the DQOs documented in the QAPP. Corrective actions may improve data quality and reduce uncertainty, and may eliminate the need to qualify or reject data.

9.3.2.1 Data Qualifiers

Qualified data are any data that have been modified or adjusted as part of statistical or mathematical evaluation, data validation, or data verification operations (ASQC 1995). Data may be qualified or rejected as a result of data validation or data verification activities. Data qualifier codes or flags are often used to identify data that has been qualified. Any scheme used should be fully explained in the QAPP and survey documentation. The following are examples of data qualifier codes or flags derived from national qualifiers assigned to results in the contract laboratory program (CLP; EPA 1994g).

U or <MDC — The radionuclide of interest was analyzed for, but the radionuclide concentration was below the minimum detectable concentration (MDC). Section 2.3.5 recommends that the actual result of the analysis be reported so this qualifier would inform the reader that the result reported is also below the MDC.

J — The associated value reported is a modified, adjusted, or estimated quantity. This qualifier might be used to identify results based on surrogate measurements (see Section 4.3.2) or gross activity measurements (*e.g.*, gross alpha, gross beta). The implication of this qualifier is that the estimate may be inaccurate or imprecise which might mean the result is inappropriate for the statistical evaluation of the results. Surrogate measurements that are not inaccurate or imprecise may or may not be associated with this qualifier. It is recommended that the potential uncertainties associated with surrogate or gross measurements be quantified and included with the results.

R — The associated value reported is unusable. The result is rejected due to serious analytical deficiencies or quality control results. These data would be rejected because they do not meet the data quality objectives of the survey.

O — The associated value reported was determined to be an outlier.

9.3.2.2 Data Validation Descriptors

Data validation is often defined by six data descriptors. These six data descriptors are summarized in Table 9.3 and discussed in detail in Appendix N. The decision maker or reviewer examines the data, documentation, and reports for each of the six data descriptors to determine if performance is within the limits specified in the DQOs during planning. The data validation process for each data descriptor should be conducted according to procedures documented in the QAPP.

Table 9.3 Suggested Content or Consideration, Impact if Not Met, and Corrective Actions for Data Descriptors

Data Descriptor	Suggested Content or Consideration	Impact if Not Met	Corrective Action
Reports to Decision Maker	• Site description • Survey design with measurement locations • Analytical method and detection limit • Detection limits (MDCs) • Background radiation data • Results on per measurement basis, qualified for analytical limitations • Field conditions for media and environment • Preliminary reports • Meteorological data, if indicated by DQOs • Field reports	• Unable to perform a quantitative radiation survey and site investigation	• Request missing information • Perform qualitative or semi-quantitative site investigation
Documentation	• Chain-of-custody records • SOPs • Field and analytical records • Measurement results related to geographic location	• Unable to identify appropriate concentration for survey unit measurements • Unable to have adequate assurance of measurement results	• Request that locations be identified • Resurveying or resampling • Correct deficiencies
Data Sources	• Historical data used meets DQO's	• Potential for Type I and Type II decision errors • Lower confidence of data quality	• Resurveying, resampling, or reanalysis for unsuitable or questionable measurements

Table 9.3 (continued)

Data Descriptor	Suggested Content or Consideration	Impact if Not Met	Corrective Action
Analytical Method and Detection Limit	• Routine methods used to analyze radionuclides of potential concern	• Unquantified precision and accuracy • Potential for Type I and Type II decision errors	• Reanalysis • Resurveying, resampling, or reanalysis • Documented statements of limitation
Data Review	• Defined level of data review for all data	• Potential for Type I and Type II decision errors • Increased variability and bias due to analytical process, calculation errors, or transcription errors	• Perform data review
Data Quality Indicators	• Surveying and sampling variability identified for each radionuclide • QC measurements to identify and quantify precision and accuracy • Surveying, sampling, and analytical precision and accuracy quantified	• Unable to quantify levels for uncertainty • Potential for Type I and Type II decision errors	• Resurveying or resampling • Perform qualitative site investigation • Documented discussion of potential limitations

Data collected should meet performance objectives for each data descriptor. If they do not, deviations should be noted and any necessary corrective action performed. Corrective action should be taken to improve data usability when performance fails to meet objectives.

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